SCIENTIFIC CRITERIA
DOCUMENT FOR STANDARD
DEVELOPMENT No. 4-84

POLYCHLORINATED DIBENZO-p-DIOXINS (PCDDs)

AND

POLYCHLORINATED DIBENZOFURANS (PCDFs)

SCIENTIFIC CRITERIA DOCUMENT
FOR STANDARD DEVELOPMENT No. 4 - 84

POLYCHLORINATED DIBENZO-p-DIOXINS (PCDDs)
AND POLYCHLORINATED DIBENZOFURANS (PCDFs)

Prepared for
Intergovernmental Relations and
Hazardous Contaminants Coordination Branch
Ontario Ministry of the Environment

September 1985

Second Printing -March 1986

PREFACE

The Hazardous Contaminants and Standards Branch has been assigned the specific mandate of co-ordinating the development of standards (guidelines) for the regulation of various hazardous substances in the environment. These standards must address both human health concerns and the protection of the environment.

A priority list of candidate substances for standard setting was developed by the Ministry of the Environment. This list includes polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs) because of the potential health hazard posed by the presence of these chemicals in the environment.

This document provides the scientific basis for the development of Provincial standards. Following input of public opinion and legal, policy and economic considerations (risk assessment) the Ministry will establish the actual standards.

In April of 1983, a Standards Co-ordinator was appointed to:

a) review existing Ministry guidelines for these substances and,

b) develop the scientific criteria for new standards which would be more closely allied to human health considerations.

Internal and External Expert Committees were formed to provide the Co-ordinator with technical direction and a forum for critical review of the document. The members of the Internal Expert Committee prepared sections of this document relevant to their specific areas of expertise. The External Committee, the Ontario Scientific Advisory Committee on Dioxins and Furans (OSAC) an international panel of experts was appointed by the Minister in June 1983 and provided technical direction and critical review.

The committees include the following members:

Internal Expert Committee

Dr. B. Birmingham*	Ministry of the Environment
Dr. R. Clement	Ministry of the Environment
Dr. D. Harding	Ministry of Labour
Mr. R. Pearson	Ministry of the Environment
Dr. D. Rokosh	Ministry of the Environment
Mr. W. Smithies	Ministry of the Environment
Dr. A. Szakolcai	Ministry of the Environment
Ms. H. Tosine	Ministry of the Environment
Mr. D. Wells	Ministry of the Environment
Ms. B. Hanna Thorpe**	Ministry of the Environment

External Expert Committee (OSAC)

Prof. O. Hutzinger	University of Bayreuth, Germany
Prof. G.L. Plaa	Universite de Montreal
Prof. S. Safe	Texas A. and M. University
Dr. E.Y. Spencer*	University of Western Ontario
Dr. B. Birmingham**	Ministry of the Environment

* Chairman

** Secretary

Acknowledgements

The advice and critical review of this document, provided by these Committee members, is gratefully acknowledged by the Ministry.

This document was typed by the Information Processing Center, Hazardous Contaminants and Standards Branch.

APPROACH TO DEVELOPING SCIENTIFIC CRITERIA

The Ministry of the Environment's approach to the development of this scientific criteria document is based upon a review of the technical literature published on the polychlorinated dibenzo-p-dioxins (PCDDs) and polychlorinated dibenzofurans (PCDFs). All relevant reports and literature were reviewed regardless of apparent quality since the aim of the Committee was to obtain an overall balanced account of the information available.

Estimates and identification of potential or actual concentrations and sources in the province of Ontario are based firstly on data collected in Ontario. This data base is one of the most extensive sets of monitoring data on these chemicals available in the world. Secondly, data from elsewhere in Canada are used and lastly, world data are used where data from the preceeding two sources are not available.

Chapter 3 deals with a review and analysis of available toxicological data on the effects of PCDDs and PCDFs. This chapter covers mammalian toxicology, genetic toxicology, carcinogenicity and effects on reproduction, human health effects and environmental toxicology and the use of the toxic equivalents approach for PCDDs and PCDFs leading to the estimation of the recommended maximum allowable daily intake of PCDDs and PCDFs.

Chapter 4 identifies sources/inputs, distribution, and fate of PCDDs and PCDFs in the environment especially of Ontario.

Chapter 5 attempts to evaluate the exposure risk to the population based on measured and estimated concentrations of PCDDs and PCDFs using the toxic equivalents approach in light of the recommended maximum allowable daily intake.

Chapter 6 reviews the current analytical methods and limitations to monitoring PCDDs and PCDFs. Recommendations for research and development to facilitate monitoring of various environmental media and biological effects are given.

TERMS OF REFERENCE

INTERNAL EXPERT COMMITTEE

(1) Identify the sources of chlorinated dioxins and furans which may contaminate air, water or land in Ontario.

(2) Review the partitioning, transport and environmental persistence and fate of chlorinated dioxins and furans and their breakdown products in air, water and soil.

(3) Estimate the potential or actual exposure of both Ontario citizens' and their environment following the dispersion of chlorinated dioxins and furans into the various environmental compartments.

(4) Review the toxicology of the various chlorinated dioxins and furans.

(5) Assess the potential adverse health effects resulting from exposure of Ontario citizens' and their environment to chlorinated dioxins and furans.

(6) Review sampling, testing and interpretation requirements for monitoring and applying abatement procedures to control the presence of chlorinated dioxins and furans in air, water and soil.

(7) Propose environmental standards, maximum acceptable concentrations (M.A.C.'s) or codes of practice to protect the health of Ontario citizens and their environment. These proposed regulations will be

based on a consideration of all routes of human and environmental exposure (including diet) and supported with relevant criteria documents. *

(8) Establish liaison and cooperate with Federal Government agencies working in this field.

* NOTE: Proposals concerning occupational exposure are beyond the mandate of this Ministry, however, the margin of safety incorporated into the proposed environmental standards anticipates this route of exposure.

EXTERNAL EXPERT COMMITTEE

(1) To provide the Minister with an independent review and assessment of the results and implications of the Ministry's sampling and scientific analysis of dioxins and furans in drinking water, soil and incinerator emissions.

(2) To advise the Minister on the toxicological significance of any findings and the feasibility of, and possible approaches to, the setting of standards for the presence of dioxins and furans in drinking water, soil and air emissions.

(3) To advise scientific staff on the methodologies and protocols used in the testing program, the interpretation of the data and data validation procedures and methods.

(4) The main concern in the Ministry's sampling and analysis program will be with the presence of dioxins and dibenzofurans; however, advice may be required on other related substances.

(5) The Committee will have unrestricted access to all test data and information it needs to make its assessments.

(6) Members may be asked by the Minister to make public their scientific views and advice after any data have been validated scientifically.

TABLE OF CONTENTS

Page

Preface i

Terms of Reference v

Table of Contents viii

List of Figures vx

List of Tables xvi

1.0 SUMMARY

1.1 EXECUTIVE SUMMARY 1-1

1.2 TECHNICAL SUMMARY AND RECOMMENDATIONS 1-6

2.0 INTRODUCTION 2-1

2.1 PHYSICAL AND CHEMICAL PROPERTIES OF PCDDs AND PCDFs 2-1

2.1.1 POLYCHLORINATED DIBENZO-P-DIOXINS (PCDDs) 2-1

2.1.2 POLYCHLORINATED DIBENZOFURANS (PCDFs) 2-6

2.2 REVIEW OF MAJOR REPORTS ON PCDDs AND PCDFs 2-6

2.3 ABBREVIATIONS 2-8

2.4 GLOSSARY 2-10

TABLE OF CONTENTS (cont'd)

Page

3.0 TOXICOLOGICAL ASSESSMENT 3-1

3.1 INTRODUCTION 3-1

3.2 MAMMALIAN TOXICOLOGY OF 2,3,7,8-T_4CDD

3.2.1 INTRODUCTION 3-6

3.2.2 ACUTE TOXICITY 3-8

3.2.3 PATHOLOGICAL AND PHYSIOLOGICAL EFFECTS 3-11

3.2.4 BIOCHEMICAL EFFECTS 3-13
3.2.4.1 Enzyme Induction 3-13
3.2.4.2 Clinical Biochemistry 3-15

3.2.5 IMMUNOTOXICITY 3-16
3.2.5.1 Immunosuppressive Effects of 2,3,7,8-T_4CDD 3-17

3.2.6 MATRIX EFFECTS ON UPTAKE (BIO-AVAILABILITY) 3-19

3.2.7 ABSORPTION, DISTRIBUTION AND EXCRETION (PHARMACOKINETICS) 3-28

3.2.8 SUMMARY 3-33

3.3 GENETIC TOXICOLOGY, CARCINOGENICITY AND EFFECTS ON REPRODUCTION OF 2,3,7,8-T_4CDD 3-35

3.3.1 GENETIC TOXICOLOGY 3-35
3.3.1.1 Salmonella/Mammalian Microsomal Mutagenicity Tests 3-35
3.3.1.2 Other Bacterial Genotoxicity Tests 3-38
3.3.1.3 Yeast Mutagenicity and Gene Conversion Assays 3-39
3.3.1.4 Summary of Microbial Mutagenicity Tests 3-39
3.3.1.5 In vitro Mammalian Cell Mutagenicity Testing 3-40
3.3.1.6 Formation of Nucleic Acid Adducts by 2,3,7,8-T_4CDD 3-44
3.3.1.7 In vivo Mammalian Mutagenicity Tests of T_4CDD 3-47

3.3.2 CARCINOGENICITY 3-49

3.3.3 TERATOGENICITY AND FETOTOXICITY 3-68

TABLE OF CONTENTS (cont'd)

Page

3.4 HUMAN HEALTH EFFECTS OF 2,3,7,8-T_4CDD 3-73

3.5 ENVIRONMENTAL TOXICOLOGY OF PCDDs AND PCDFs 3-81

3.5.1 TERRESTRIAL TOXICOLOGY 3-82
3.5.1.1 Toxicity to Wild and Domestic Birds 3-82
3.5.1.2 Toxicity to Wild and Domestic Animals 3-87
3.5.1.3 Toxicity to Natural Vegetation and Field Crops 3-96

3.5.2 AQUATIC TOXICOLOGY 3-97
3.5.2.1 Toxicity to Fish 3-97
3.5.2.2 Toxicity to Aquatic Invertebrates 3-99

3.6 COMPARATIVE TOXICOLOGY OF PCDDs AND PCDFs 3-102

3.6.1 COMPARATIVE TOXICITY OF PCDD CONGENERS 3-102

3.6.2 ACUTE TOXICITY OF PCDDs 3-102

3.6.3 BIOCHEMICAL RESPONSES OF PCDDs 3-105

3.6.4 TERATOLOGY OF PCDDs 3-107

3.6.5 CARCINOGENICITY 3-110

3.6.6 COMPARATIVE TOXICITY OF PCDFs 3-111

3.6.7 2,3,7,8-T_4CDD TOXIC EQUIVALENTS 3-117

3.7 ESTIMATION OF MAXIMUM ALLOWABLE DAILY INTAKE 3-130

3.7.1 INTRODUCTION (SUMMARY OF HAZARD IDENTIFICATION AND SELECTION OF CRITICAL TOXICOLOGICAL CRITERIA) 3-130

3.7.2 HAZARD ESTIMATION 3-138

3.7.3 LOW DOSE EXTRAPOLATION 3-142

3.7.4 ESTIMATION OF MAXIMUM ALLOWABLE DAILY INTAKE 3-144

TABLE OF CONTENTS (cont'd)

Page

4.0 SOURCES/INPUTS, DISTRIBUTION AND FATE OF PCDDs AND PCDFs IN THE ENVIRONMENT

4.1 INTRODUCTION 4-1

4.2 SOURCES/INPUTS 4-3

4.2.1 COMBUSTION SOURCES 4-3
4.2.1.1 Municipal Waste Incineration 4-3
4.2.1.2 Chemical Waste Incineration 4-45
4.2.1.3 Biological Waste Incineration 4-53
4.2.1.4 Wood Combustion 4-54
4.2.1.5 Fossil Fuel Combustion For Power Generation 4-64
4.2.1.6 Other Combustion Sources 4-67
4.2.1.7 Estimates of Atmospheric Input to the Ontario Environment from Combustion Sources 4-71

4.2.2 CHEMICAL MANUFACTURING AND USE 4-81
4.2.2.1 Chlorophenol Production and Use 4-82
4.2.2.2 Herbicide Production/Formulation/Use 4-89
4.2.2.3 Previous Herbicide Manufacture and Use 4-92
4.2.2.4 PCB Manufacture and Use 4-93
4.2.2.5 Commercial/Domestic Products/Use 4-95

4.2.3 WASTE DISPOSAL SITES 4-96

4.2.3.1 Elmira 4-98
4.2.3.2 Wood Preservation Sites 4-101
4.2.3.3 Landfill Sites Accepting Chemicals and Other Industrial Sites Potentially Contaminated with PCDDs and PCDFs 4-103
4.2.3.4 Incinerator Ash Disposal Sites 4-104
4.2.3.5 Municipal/Industrial Sewage Sludge Disposal 4-106

TABLE OF CONTENTS (cont'd)

Page

4.2.4 "NIAGARA FRONTIER" AND OTHER TRANS-BOUNDARY SOURCES 4-108
- 4.2.4.1 Niagara Frontier 4-108
- 4.2.4.2 Dow, Midland, Michigan 4-110
- 4.2.4.3 St. Clair River/Detroit River 4-111
- 4.2.4.4 Ontario Water Utilities 4-112

4.2.5 SUMMARY OF PCDDs AND PCDFs ENTERING THE ONTARIO ENVIRONMENT 4-116

4.3 ENVIRONMENTAL TRANSPORT, DISTRIBUTION, AND FATE 4-124

4.3.1 ATMOSPHERIC TRANSPORT 4-124
- 4.3.1.1 Long Range Transport in the Atmosphere 4-124
- 4.3.1.2 Chemical Stability in the Atmosphere 4-124
- 4.3.1.3 Atmospheric Monitoring 4-125

4.3.2 DEPOSITION, FATE, AND MOBILITY IN THE TERRESTRIAL ENVIRONMENT 4-125
- 4.3.2.1 Deposition to Soil and Plants from Atmospheric Sources 4-126
- 4.3.2.2 Photochemical Synthesis 4-140
- 4.3.2.3 Degradation in Soils and Plants 4-142
- 4.3.2.4 Physical Transport in Soils 4-148

4.3.3 AQUATIC FATE AND PERSISTENCE 4-152
- 4.3.3.1 Non-Biological Processes 4-152
- 4.3.3.2 Biological Processes 4-155

4.3.4 BIOLOGICAL ACCUMULATION/MAGNIFICATION 4-158
- 4.3.4.1 Wildlife and Domestic Animals 4-158
- 4.3.4.2 Vegetation 4-163
- 4.3.4.3 Aquatic Biota 4-167
- 4.3.4.4 Residue Levels in Fish From Ontario Waters 4-171
- 4.3.4.5 Residue Levels of PCDDs and PCDFs in Humans 4-177

TABLE OF CONTENTS (cont'd)

Page

5.0 EXPOSURE ASSESSMENT

5.1 INTRODUCTION 5-1

5.2 EXPOSURE PATHWAYS 5-5

5.2.1 AIR 5-5

5.2.2 WATER 5-7

5.2.3 SOIL 5-10

5.2.4 FOOD 5-11
5.2.4.1 Fish 5-11
5.2.4.2 Other Food 5-12

5.2.5 SUMMARY 5-13

5.3 ESTIMATED DOSES 5-14

5.3.1 DAILY INHALATION 5-16

5.3.2 DAILY INGESTION FROM WATER 5-17

5.3.3 DAILY PICA INGESTION FROM SOIL (CHILDREN) 5-18

5.3.4 DAILY DERMAL 5-19

5.3.5 DAILY INGESTION FROM FOOD 5-20

5.3.6 SUMMARY 5-21

5.4 EXPOSURE SCENARIOS, SPECIAL GROUPS AT RISK AND CURRENT LEVELS OF EXPOSURE 5-22

5.4.1 EXPOSURE SCENARIOS, SPECIAL GROUPS AT RISK AND CURRENT LEVELS OF EXPOSURE 5-22

5.4.2 SITE-SPECIFIC EXPOSURE ASSESSMENT 5-27

5.4.3 IMPLICATIONS FOR RISK MANAGEMENT 5-29

TABLE OF CONTENTS (cont'd)

Page

6.0 ANALYTICAL METHODS FOR MONITORING PCDDs AND PCDFs 6-1

6.1 PCDDs AND PCDFs: REVIEW OF ANALYTICAL METHODS 6-1

6.2 BIOLOGICAL ANALYSIS 6-16

6.3 CONCLUSIONS AND RECOMMENDATIONS ON CURRENT LIMITATIONS AND FUTURE RESEARCH NEEDS 6-19

APPENDICES

Appendix I Estimation of Current Levels of Exposure from Adipose Tissue Levels in Humans I-1

Appendix II Tables 3.2.6B to Table 3.2.6D Bioavailability Related Studies II-1

Appendix III A possible Dioxin Standard Partitioning Algorithm III-1

BIBLIOGRAPHY

LIST OF FIGURES

Page

Figure 3.5.2.1A PCDD Concentration/Effect Data on Aquatic Organisms 3-100

Figure 4.2.1.1A Distribution of PCDDs and PCDFs in Precipitated Fly Ash in Municipal Incinerators 4-22

LIST OF TABLES

Page

Table 2.1.1A A List of PCDD Congeners indicating Numbers of Isomers in each Chlorine Substitution Group 2-2

Table 2.1.1B Physical Properties of Selected PCDDs 2-3

Table 2.1.2A A List of PCDF Congeners indicating Numbers of Isomers in each Chlorine Substitution Group 2-4

Table 3.2.2A Acute Mammalian Toxicity of 2,3,7,8-T_4CDD 3-9

Table 3.2.2B Acute toxicity and Histopathology of 2,3,7,8-T_4CDD or Related Chlorinated Aromatic Hydrocarbons: Species Differences 3-10

Table 3.2.5.1A Effect of 2,3,7,8-T_4CDD Treatment on Mortality Following Challenge with Herpes Virus II 3-18

Table 3.2.6A Summary of Absorption Studies 3-26

Table 3.2.7A Pharmacokinetic Processes and their Relevance to Toxicological Assessment 3-28

Table 3.2.7B Rates of Elimination of 2,3,7,8-T_4CDD or 2,3,7,8-T_4CDF in Various Species 3-30

Table 3.2.7C Summary of Pharmacokinetic Estimates 3-34

Table 3.3.1.1A Salmonella Mammalian Microsomal Mutagenicity Test Results 3-36

Table 3.3.1.5A Mutagenicity and Toxicity of PCDDs in Baby Hamster Kidney Cells (BHK 21 C13) 3-42

Table 3.3.1.7A In Vivo Genotoxic Events associated with 2,3,7,8-T_4CDD 3-45

Table 3.3.2A Summary of Neoplastic Alterations observed in Rats Fed Subacute Levels of 2,3,7,8-T_4CDD for 78 weeks 3-50

Table 3.3.2B Summary of Significant Tumour Incidence in Male Sprague-Dawley Rats Maintained on Diets containing 2,3,7,8-T_4CDD 3-51

Table 3.3.2C Summary of Significant Tumour Incidence in Female Sprague-Dawley Rats Maintained on Diets containing 2,3,7,8-T_4CDD 3-53

LIST OF TABLES (con't)

		Page
Table 3.3.2D	Summary of Significant Tumour Numbers in Male Osborne-Mendel Rats Administered 2,3,7,8-T_4CDD by Gavage	3-56
Table 3.3.2E	Summary of Significant Tumour Numbers in Female Osborne-Mendel Rats Administered with 2,3,7,8-T_4CDD by Gavage	3-57
Table 3.3.2F	Summary of Number of Significant Tumours in Male B5C3F1 Mice Administered 2,3,7,8-T_4CDD by Gavage	3-60
Table 3.3.2G	Summary of Numbers of Significant Tumours in Female B6C3F1 Mice Administered 2,3,7,8-T_4CDD by Gavage	3-61
Table 3.3.3A	Summary of Teratogenicity and Fetotoxicity of 2,3,7,8-T_4CDD in Rats and Mice	3-69
Table 3.4A	Rosults of Case-Control Studies Carried out in Sweden	3-74
Table 3.4B	Toxic Effects of 2,3,7,8-T_4CDD in Man	3-76
Table 3.4C	Carcinogenicity and Epidemiology Studies Associated with PCDDs	3-77
Table 3.5.1.1A	Effects of T_4CDD on Wild and Domestic Bird Species	3-85
Table 3.6.2A	Acute Toxicities and AHH Induction Efficiencies of PCDD	3-104
Table 3.6.4A	Comparative Teratology of PCDD Isomers in Rats	3-109
Table 3.6.5A	Summary of Data from Animal Carcinogenicity Bioassays of PCDDs	3-110
Table 3.6.6A	Hepatic Aryl Hydrocarbon Hydroxylase Induction Efficiency by PCDF Isomers	3-113
Table 3.6.7A	2,3,7,8-T_4CDD Toxic Equivalents	3-118
Table 3.6.7B	Options for Prorating the Proposed Standard	3-122

LIST OF TABLES (con't)

		Page
Table 3.6.7C	Estimated Relative Toxicity of PCDD and PCDF Isomers to 2,3,7,8-T_4CDD	3-129
Table 3.7.2A	Lower Response Limits for 2,3,7,8-T_4CDD in Rodents	3-139
Table 3.7.3A	Estimates and Approximate Lower Confidence Limits for the Virtually Safe Dose (VSD) of 2,3,7,8-T_4CDD at Various Risk Levels	3-144
Table 4.2.1.1A	Levels of PCDD/PCDF in Precipitated Fly Ash from Municipal Waste Incinerator Plants	4-13
Table 4.2.1.1B	Levels of PCDD/PCDF in Stack-Collected Particulates from Municipal Waste Incinerator Plants	4-16
Table 4.2.1.1C	Levels of PCDD/PCDF in Stack Emissions from Municipal Waste Incinerators	4-17
Table 4.2.1.1D	Summary of Average PCDD and PCDF Congener Concentrations and Calculated Total (T_4 - O_8) Values for Municipal Waste Incinerator Emissions	4-23
Table 4.2.1.1E	Average Percent Composition of PCDDs and PCDFs in Emissions from Municipal Waste Incinerators	4-24
Table 4.2.1.1F	Variation in PCDD and PCDF Concentration in Precipitated Fly Ash, Stack-Collected Particulates and Total Stack Emissions from Samples Collected over Different Time Intervals from Municipal Waste Incinerators	4-25
Table 4.2.1.1G	Size Fractionated PCDD Concentration in Fly Ash from Municipal Waste Incinerators	4-29
Table 4.2.1.1H	Number of Isomers Detected in PCDD and PCDF Chlorinated Congeners in Emissions from Municipal Waste Incinerators	4-35
Table 4.2.1.1I	Isomer Specific Analyses Showing Percentage of Total T_4CDD Measured in Municipal Waste Incinerator Emissions	4-36

LIST OF TABLES (con't)

		Page
Table 4.2.1.1J	Average Isomer Composition (%) of Total T_4CDDs in Fly Ash and Stack Emissions	4-37
Table 4.2.1.1K	Dispersion Model Predictions of Ground Level Concentration in the Vicinity of Municipal Waste Incinerators	4-38
Table 4.2.1.1L	Estimated Total Annual Output of PCDD and PCDF from Municipal Waste Incinerators (Excluding Ontario)	4-39
Table 4.2.1.2A	PCDD and PCDF Concentrations in Combustion Effluents from Industrial Chemical Waste Incineration	4-47
Table 4.2.1.3A	Concentration of PCDDs and PCDFs in Stack Emissions from Royale Jubilee Hospital Incinerator - Victoria, B.C. - 1983	4-53
Table 4.2.1.4A	PCDD and PCDF Concentrations in Combustion Effluents from the Combustion of Chemically Treated Wood/Wood Products	4-57
Table 4.2.1.4B	PCDD and PCDF Concentrations in Effluents from the Combustion of Untreated Wood	4-62
Table 4.2.1.5A	PCDD and PCDF Concentrations in Effluents from Fossil Fuel Combustion	4-65
Table 4.2.1.5B	Total Particulate - Phase Stack Emission Concentrations of PCDDs and PCDFs from Two Canadian Coal-Fired Sources	4-66
Table 4.2.1.6A	Estimates of Chlorinated Dibenzofurans from Toronto Transformer Fire	4-69
Table 4.2.1.7A	Calculated Annual Emissions of PCDD and PCDF from Two Municipal Waste and One Sewage Sludge Incinerator in Ontario	4-74
Table 4.2.1.7B	Maximum Annual Average Concentration near Incinerators in Ontario	4-76
Table 4.2.1.7C	Calculated Average Annual Output of PCDD and PCDF Congeners from SWARU, Hamilton	4-77
Table 4.2.2A	PCDF Concentrations in Aroclor, Clophen and Phenoclor	4-82

LIST OF TABLES (con't)

		Page
Table 4.2.2.1A	Estimate of T_4CDD - O_8CDD Contamination in PCP Used in Ontario Wood Preservation Plants	4-87
Table 4.2.2.2A	Levels of D_2-, T_3- and T_4CDDs in 2,4-D Formulations in Canada	4-90
Table 4.2.2.2B	Levels of D_2-, T_3- and T_4CDDs in 2,4-D Formulations in Canada in 1983	4-91
Table 4.2.3.1A	Uniroyal, Elmira Water Analyses (Test Well 18S)	4-100
Table 4.2.3.4A	Estimates of Input of PCDD and PCDF from Incinerator Ash to Landfill Sites	4-105
Table 4.2.3.5A	Preliminary Monitoring Results for Levels of PCDD/PCDF in Hamilton STP (Environment Canada)	4-107
Table 4.2.3.5B	Preliminary Results for levels of PCDD/PCDF in Elmira STP Sludge (MOE)	4-108
Table 4.2.4.1A	Concentrations of PCDDs and PCDFs in Niagara River Water near the Hyde Park Dump Site, Niagara Falls, N.Y. 1983	4-110
Table 4.2.4.2A	Levels of T_4CDD in Fish from Tittabawassee River and Saginaw Bay, Lake Huron (near Dow)	4-111
Table 4.2.4.4A	Analysis of Water Samples for Total 2,3,7,8-T_4CDD January 1980 - January 1981	4-113
Table 4.2.4.4B	Analysis of Water Samples for Total 2,3,7,8-T_4CDD (Western Lake Ontario) June 1980 - September 1982	4-114
Table 4.2.4.4C	Analysis of Water Samples for Total 2,3,7,8-T_4CDD (Western Lake Ontario) September 1981 - September 1982	4-114
Table 4.2.4.4D	Analysis of Water Samples for Total T_4CDD (Western Lake Ontario) Comparison of Ministry of the Environment and Health and Welfare, Canada Data	4-115
Table 4.2.5A	Summary of Estimated Quantities of PCDDs and PCDFs Entering the Ontario Environment	4-119

LIST OF TABLES (con't)

		Page
Table 4.2.5B	Summary of Estimated Quantities of PCDDs and PCDFs Entering the Ontario Environment as 2,3,7,8-T_4CDD Toxic Equivalents	4-123
Table 4.3.1.3A	Atmospheric Monitoring Results	4-127
Table 4.3.2.1A	Summary of PCDD and PCDF Levels in Soils	4-129
Table 4.3.2.1B	PCDD and PCDF Concentration in Surface soil Collected in the Vicinity of a Municipal Refuse Incinerator, Hamilton, Ontario, 1983	4-136
Table 4.3.2.1C	T_4CDD Analysis of Grass Samples Collected in a Zone Extending up to 900 m SE of the TCP Factory, Seveso, Italy	4-139
Table 4.3.4.2A	T_4CDD Concentrations in Vegetation Grown Close to the Ground in Zones B and R, Seveso, Italy, 1977	4-166
Table 4.3.4.2B	Plant Uptake Data for Perennial Grasses and Broadleaf Plants from Studies Conducted on Selected Sites of Grid 1, Eglin A.F.B.	4-166
Table 4.3.4.3A	Biological Accumulation of TCDD in Aquatic Organisms	4-168
Table 4.3.4.3B	PCDF Distribution in a Pond Known to be Contaminated by Aroclor 1260	4-170
Table 4.3.4.4A	2,3,7,8-T_4CDD Identified in Great Lakes Fish	4-173
Table 4.3.4.4B	1980 Survey of Ontario resident Sport Fishermen Numbers (1,000's) of Fish Caught and Kept by Adults on Selected Water Bodies	4-176
Table 4.3.4.5A	TCDD Levels in Human Adipose Tissue	4-180
Table 5.2.1A	Conversion of Total PCDD and PCDF Residues in Air Near the Zone of Maximum Exposure near SWARU to 2,3,7,8-T_4CDD Equivalents	5-7
Table 5.2.2A	Conversion of PCDD and PCDF Residues in Raw Water to 2,3,7,8-T_4CDD Equivalents	5-9

LIST OF TABLES (con't)

		Page
Table 5.2.3A	Conversion of Total PCDD and PCDF Residues in Urban Ontario Soil to 2,3,7,8-T_4CDD Equivalents	5-11
Table 5.2.4A	Conversion of PCDD and PCDF Residues in Selected Food Items to 2,3,7,8-T_4CDD Toxic Equivalents	5-12
Table 5.2.5A	Recommended Maximum Daily Intake of 2,3,7,8-T_4CDD or its Equivalent of PCDDs and PCDFs	5-15
Table 5.3.1A	Hypothetical Daily Dose of 2,3,7,8-T_4CDD or its Toxic Equivalent from Inhalation	5-17
Table 5.3.2A	Hypothetical Daily Dose of 2,3,7,8-T_4CDD or its Toxic Equivalent from Water	5-18
Table 5.3.3A	Hypothetical Dose of 2,3,7,8-T_4CDD or its Toxic Equivalent from Daily Ingestion of Soil-Bound PCDDs and PCDFs by Children	5-19
Table 5.3.4A	Hypothetical Dose of 2,3,7,8-T_4CDD or its Toxic Equivalent from Daily Dermal Absorption	5-20
Table 5.3.5A	Hypothetical Dose of 2,3,7,8-T_4CDD or its Equivalent from Daily Ingestion of Selected Food Items	5-20
Table 5.4.1A	Pathways, Special Groups and Sites of Contamination as They Relate to Human Exposure	5-22
Table 5.4.1B	Summary of Exposure Assessment (Worst Case)	5-24
Table 5.4.1C	Estimation of Current Levels of Exposure	5-26
Table 5.4.2A	Guidelines for Site-Specific Exposure Assessment	5-28
Table 6.1A	Relative GC-MS Response Factors for PCDF Compounds: Normalized to 2,3,7,8-T_4CDD = 100	6-13

LIST OF TABLES (con't)

		Page
Appendix II -	Tables 3.2.6B to Table 3.2.6D Bioavailability Related Studies	II-1

1.0 SUMMARY

1.1 EXECUTIVE SUMMARY

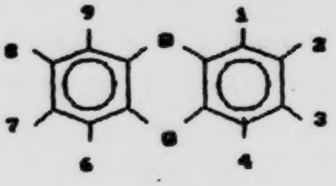
The terms dioxin and furan refer to families of 75 related chemical compounds known as polychlorinated dibenzo-p-dioxins (PCDDs) and 135 related chemical compounds known as polychlorinated dibenzofurans (PCDFs) respectively. These compounds are not intentionally made for any purpose; they are unavoidable by-products created in the manufacture of other chemicals such as some pesticides, or as a result of incomplete combustion of mixtures containing chlorine atoms and organic compounds.

These two families of compounds possess similar chemical structures, patterns of toxic and biological responses and may share a common mechanism of action at the cellular level. Therefore they are being dealt with as a group for the purposes of development of environmental standards.

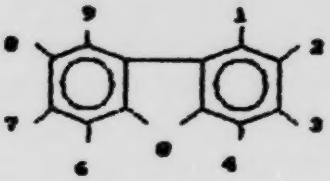
PCDDs are composed of a triple-ring structure made up of two benzene rings connected to each other by a pair of oxygen atoms. The 75 PCDDs differ only by the number and position of the chlorine atoms attached to the benzene rings.

The PCDF molecules are also composed of a triple-ring structure involving two benzene rings connected both directly to each other and by a single oxygen atom. The 135 PCDFs differ also by the number and position of the chlorine atoms attached to the benzene rings.

For both PCDDs and PCDFs the positions of the chlorine atoms on the molecule determine the toxicity of the specific isomer. The most toxic forms of PCDDs and PCDFs are those containing 4-6 chlorine atoms, with four of the chlorine atoms at the lateral positions, i.e., 2, 3, 7 and 8.



PCDD



PCDF

The 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-T_4CDD) is the most toxic of all the PCDDs and PCDFs. It is also the form generally assumed by the public to be present when the term dioxin is used.

In Ontario, there is no current chemical manufacturing of the 2,4,5-trichlorophenol, nor formulation of 2,4,5-T and 2,4-D herbicides or pentachlorophenol or hexachlorophenol chemicals with which PCDD and PCDF contamination has been associated. Current sources of PCDDs and PCDFs in the Ontario environment are from incineration processes or the use of products which contain trace amounts of PCDDs and PCDFs.

The PCDDs and PCDFs from these sources are usually complex mixtures. The 2,3,7,8-T_4CDD isomer is generally only a small percent of the total PCDDs and PCDFs present. This is in contrast to the problem in the United States where because of extensive chemical manufacturing and waste disposal, 2,3,7,8-T_4CDD is a serious environmental contaminant.

Analyses carried out by National Health and Welfare Canada on PCDD and PCDF residues in tissues of deceased and living persons, indicate body burdens of some PCDD and PCDF isomers in the majority of the samples analyzed.

These results suggest that PCDDs and PCDFs are ubiquitous at low levels in the Ontario environment.

Recommendations and Conclusions

Based on extensive reviews of the literature on the toxicology of PCDDs and PCDFs, the following conclusions and recommendations have been reached:

(a) Sources and Exposure

The document reviews extensively the current sources of PCDDs and PCDFs in Ontario based firstly upon analytical data from Ontario and where this is absent, upon extrapolation from other Canadian or international data. In order of decreasing contribution to the Ontario environment, the sources have been identified as:

i) combustion sources including municipal refuse and sewage sludge incineration;

ii) use of chemical products such as chlorinated phenols; and,
iii) other sources such as transboundary water and air contamination, chemical wastes, commercial and domestic wastes, polychlorinated biphenyls (PCBs) and sewage.

Based on preliminary exposure assessment the major routes of exposure in order of decreasing contribution appear to be;

i) ambient air in the vicinity of incineration sources;
ii) diet, especially some sport fish from Lake Ontario;
iii) soil; and,
iv) surface water - it should be noted that no PCDDs or PCDFs have been found in samples of finished drinking water in Ontario.

(b) PCDD and PCDF Toxic Equivalents

Based on limited acute animal toxicity studies and biological analysis, the toxicity of the various PCDD and PCDF isomers is extremely variable. The toxicity of the various isomers vary by factors of up to 10,000 with the congeners with 8 chlorine atoms (octa) being among the least toxic. The most toxic isomer is 2,3,7,8-T_4CDD. Based on analyses done in Ontario, 2,3,7,8-T_4CDD comprises a very small percentage of the PCDD and PCDF isomers found in environmental samples with the exception of some sport fish from the Great Lakes especially the western-end of Lake Ontario.

Therefore it is recommended that the development of environmental standards for PCDDs and PCDFs in Ontario should be based on a toxic equivalent approach with 2,3,7,8-T_4CDD being the basis for comparison. Numerical conversion factors to convert the concentrations of other PCDD and PCDF congeners to equivalent concentrations of 2,3,7,8-T_4CDD which would exhibit similar toxicity are included (Section 3.6.7).

(c) Recommended Maximum Daily Intake

Based on reliable chronic animal studies and extensive but inconclusive human epidemiological data, it is recommended that a threshold-safety factor approach be used to develop a recommended allowable daily intake (Section 3.7). The recommended maximum daily intake for 2,3,7,8-tetrachlorodibenzo-p-dioxin (2,3,7,8-T_4CDD) or its toxic equivalent of PCDDs and PCDFs is 10 picograms/kilogram of body weight/day for humans.

The next step in the process of developing environmental standards for PCCDs and PCDFs is the risk assessment stage. This phase integrates the science with public opinion, socio-economic considerations and abatement feasibility options to produce a final allowable daily intake and specific standards (guidelines, criteria, objectives) for the various environmental media, e.g., ambient air, drinking water, surface water, soil, etc. These specific standards should ensure that the total exposure is at or below what is determined to be the allowable dose.

1.2 TECHNICAL SUMMARY AND RECOMMENDATIONS

In Chapters 3, 4 and 5 the important conclusions and recommendations based upon the scientific analysis are found in bold faced type at the appropriate points in the text. The following is a summary of those major conclusions and recommendations organized to follow the format of Chapters 3, 4 and 5. References refer to the sections wherein the fuller discussion leading to those conclusions and recommendations can be found.

TOXICOLOGICAL ASSESSMENT

Mammalian Toxicology (3.2)

PCDDs and PCDFs share similar chemical structures, patterns of toxic and biological responses and possibly a common mechanism of action at the cellular level. While the actual mechanism of toxicity is unclear at the present time, the more toxic or biologically active members of this group of chemicals share several of the following series of responses in experimental animals:

i) a wasting syndrome
ii) skin disorders
iii) effects on the immune system
iv) impaired liver function
v) altered hematological functions
vi) impaired reproduction
vii) increased incidence of tumours
viii) induction of numerous enzymes including the hepatic cytochrome P-448 (or P-450) - dependent monooxygenases (i.e. aryl hydrocarbon hydroxylase, AHH).

Current evidence suggests that a cytosolic receptor protein plays a pivotal role in initiating the biological and toxic responses described above. Most of our knowledge is based on 2,3,7,8-T_4CDD, the most toxic and active PCDD. Consequently, this PCDD forms the major focus of this scientific criteria document and the basis of the proposed standard.

Other PCDDs and PCDFs are less toxic and less well understood and are best described in terms of their toxic equivalence to 2,3,7,8-T_4CDD.

Acute Toxicity (3.2.2)

There is wide variation in the dosage of 2,3,7,8-T_4CDD required to cause death among mammals (oral LD_{50} 0.6 - 5000 ug/kg body weight). Death of the animals does not occur for several weeks and is preceded by a wasting syndrome which includes progressive weight loss, skin disorders, and changes to the thymus, blood and liver. 2,3,7,8-T_4CDD is acutely toxic to most laboratory animals e.g. guinea pigs, rats, monkeys, rabbits and mice at extremely low doses of less than 500 ug/kg body weight.

This extreme acute toxicity of 2,3,7,8-T_4CDD should be considered when assessing its effects on carcinogenicity, reproductive functions and the immune system.

Enzyme Induction (3.2.4.1)

2,3,7,8-T_4CDD and some related PCDDs and PCDFs are potent inducers (i.e. trigger enzyme production) of several enzyme systems including diverse hepatic and extrahepatic drug-metabolizing enzymes in some animal species. The rodent hepatic cytochrome P-448 dependent monooxygenases (i.e. aryl hydrocarbon hydroxylase, AHH) are readily induced by 2,3,7,8-T_4CDD.

Induction of these enzymes is a sensitive indicator of PCDD or PCDF exposure. Enzyme induction is not a toxic response *per se* but may be used to estimate the biological activity of other PCDDs or PCDFs with reference to 2,3,7,8-T_4CDD to determine toxic equivalence.

Immunotoxicity (3.2.5)

A common feature of 2,3,7,8-T_4CDD intoxication in different animals is thymic atrophy usually associated with decreased activity and atrophy of the lymphatic tissues.

These clinical responses are probably related to the general suppression of the immune system observed following PCDD and PCDF exposure. Effects on both the cell-mediated and the humoral-mediated functions of the immune system in mice have been reported.

This work on the immunosuppressive effects of 2,3,7,8-T_4CDD is in its early stages of understanding and adverse effects on the immune system at exposure levels lower than those causing other toxic effects has not been demonstrated.

Matrix Effects on Uptake (Bio-Availability) (Section 3.2.6)

It is concluded from a review of current bio-availability studies that assuming that 100% of the PCDDs and PCDFs associated with particulate matter, i.e. soil, fly ash and sediments, which are ingested, dermally absorbed or inhaled are all biologically available, represents the "worst case" assumption. Studies show that this assumption introduces at least two- to five-fold safety factor in the case of oral exposure, into the exposure assessment.

Absorption, Distribution and Excretion (Pharmacokinetics) (Section 3.2.7)

Numerous studies have been done on the absorption, distribution, and excretion of 2,3,7,8-T_4CDD in animals. Other PCDDs and PCDFs have been studied to a much lesser extent. Research on PCDFs focuses primarily on 2,3,7,8-T_4CDF. Some evidence is available on distribution and excretion of PCDDs and PCDFs in humans following accidental or occupation exposure.

Based on studies in animals and man, the major storage sites for PCDDs and PCDFs are in the liver and adipose tissues. In rodents, the levels in the liver are higher than in adipose tissues; in primates (monkeys and man) the reverse is found.

Metabolism of 2,3,7,8-T_4CDD and 2,3,7,8-T_4CDF varies widely with the animal species studied.

Excretion rates and excretion products have been the subject of a number of studies. The half-life of 2,3,7,8-T_4CDD ranges from 11 to 365 days in various species; from 1% to 41% of 2,3,7,8-T_4CDD is excreted in urine and 59% to 99% in faeces. For 2,3,7,8- T_4CDF half-lives in different animal species range from 2 to 40 days; excretion via urine from 2% to 20% and via faeces from 5% to 82%.

In humans, estimates and measurements of half-lives for PCDDs and PCDFs equal or exceed 365 days.

Genetic Toxicology (3.3.1)

2,3,7,8-T_4CDD does not form DNA adducts, i.e. it does not chemically bind to DNA. Therefore it is unlikely to cause mutations by chemically changing DNA directly.

This compound has been extensively tested in *in vitro* microbial mutagenicity tests, such as the Ames test, with mainly negative results. More limited information is available from *in vitro* and *in vivo* mammalian test systems for mutagenicity and clastogenicity.

Conflicting or insufficient data from these mammalian test systems does not rule out the potential of 2,3,7,8-T_4CDD for mutagenic activity. However this compound has not been shown to act as a classical mutagen.

Additional research using other genetic endpoints is required to resolve the uncertainty about the mutagenic activity of 2,3,7,8-T_4CDD in mammalian test systems.

The relative potency of other PCDDs and PCDFs in *in vitro* and *in vivo* genotoxicity test systems is virtually unknown. This area of research should have high priority.

Carcinogenicity (3.3.2)

Composite data from several comprehensive carcinogenicity bioassays using rats or mice have demonstrated an association between 2,3,7,8-T_4CDD and increased incidences of specific tumour types at relatively low doses.

The compound has been demonstrated carcinogenic over the range of 0.007 to 0.3 ug/kg/day in Sprague-Dawley and Osborne-Mendel rats as well as in B6C3F1 mice when administered by oral or gavage routes. 2,3,7,8- T_4CDD was also shown to be carcinogenic in female Swiss-Webster mice chronically dosed at about 0.1 ug/kg/day by a dermal route.

Those studies involving oral administration (2,3,7,8-T_4CDD is most toxic using this route of exposure) have reliable dose-response data and indicate clear NOEL (0.001 to 0.0014 ug/kg/day) for tumour incidence.

In general, 2,3,7,8- T_4CDD administration was associated with relatively few histologically different tumour types. The onset of these tumours appeared late in the study and generally at similar time to onset of similar, but less frequent, tumors in the control animals.

Although 2,3,7,8-T_4CDD has been found to be carcinogenic in rats and mice it is difficult at this stage to predict the ability of this compound to induce tumours in humans.

Studies designed to resolve the mode of action of 2,3,7,8-T_4CDD as a carcinogen have drawn conflicting conclusions. Results of short term genotoxicity tests suggest that 2,3,7,8-T_4CDD is not genotoxic in a classical sense and does not act directly on the chromosomes or the DNA molecule itself. Consequently 2,3,7,8-T_4CDD appears to produce tumours in rodents by an indirect mechanism.

Teratogenicity and Fetotoxicity (3.3.3)

Several studies indicate that 2,3,7,8-T_4CDD is teratogenic in mice causing cleft palate in offspring of exposed females.

In rats, this compound is primarily fetotoxic producing internal hemorrhages in the fetus.

Fetotoxic kidney abnormalities occur in both rats and mice.

The NOEL (No-Observable-Effect-Level) for teratogenic or fetotoxic effects is 0.1 ug 2,3,7,8-T_4CDD/kg/day in rats and mice respectively.

Human Health Effects of 2,3,7,8-T_4CDD (3.4)

Humans have been acutely exposed to PCDDs and PCDFs in certain occupations, e.g. pesticide manufacture and use, crop protection, tanning, textiles, wood preserving and the manufacture of electrical transformers and capacitors. High levels of exposure have followed industrial accidents or episodes leading to localized contamination of air, water, soil or diet e.g. explosions at pesticide production facilities, transformer fires or contamination of cooking oil, or faulty disposal of industrial wastes leading to soil and groundwater contamination e.g. road oiling or disposal in non-secure landfill.

Acute exposure to PCDDs or PCDFs results in symptoms (e.g. chloracne) which slowly decrease over a prolonged period of time.

Epidemiological studies to date do not present sufficient evidence to establish a causal relationship between exposure to PCDDs or PCDFs and a chronic human health effect relating to carcinogenesis, coronary disease or impairment of the immune system.

A major problem which requires further investigation is accurate assessment of human exposure to PCDDs and PCDFs.

Environmental Toxicology (3.5)

In the past, domestic animals and birds have been exposed to PCDDs and PCDFs following application of some types of herbicides for crop protection or use of wood preservatives on farm structures and animal bedding materials or ingestion of contaminated soil or commercially- produced animal feed. Wildlife have been exposed following application of some herbicides to rangeland, brush or forests or following industrial accidents or military use.

Laboratory ingestion of T_4CDD-contaminated soil from Times Beach and Minker Stout Horse Arena, Missouri by test animals indicates that T_4CDD is biologically available. Acute exposure has led to several well investigated incidents (Chick oedema disease - U.S.A. - 1957; Eglin U.S. Air Force Base, Florida, 1970s; Missouri Horse Arena, 1971; Seveso - Italy - 1976; Ontario farrowing pen, 1983). Animals surviving the initial toxic episode and/or living their entire life cycle in contaminated areas display no long term toxic effects. While visible effects were observed at doses as low as 1 ug/kg/day, the dose-response data are insufficient to permit derivation of a long term no-effect concentration.

Fish and aquatic invertebrates may be exposed to PCDDs and PCDFs in surface or ground water resulting from pesticide application to cropland or forests, industrial effluents or leachate from waste disposal sites. Adverse effects have been observed in fish exposed to concentrations as low as 1 ng 2,3,7,8-T_4CDD/L usually with an extended latency period. Embryonic and larval forms of fish are most sensitive.

The phytotoxicity of PCDDs or PCDFs to natural or cultivated vegetation is unknown. No firm evidence implicates the direct phytotoxicity of these compounds to natural vegetation or field crops.

Comparative Toxicology of PCDDs and PCDFs (3.6)

Animal studies (LD_{50}, body weight gain, thymus/body weight ratio, teratology) and in vitro experiments (enzyme induction, cytosolic receptor binding), indicate that there are pronounced differences in the toxic and biological effects of the different PCDD and PCDF congeners. These differences are strongly correlated with the chlorine substitution patterns of the PCDD or PCDF molecule.

The isomers with the highest activity and acute toxicity are those with 4 to 6 chlorines and all lateral (2,3,7 and 8) positions substituted with chlorine. Further chlorine substitution or removal of lateral chlorines results in congeners with decreased toxicities or biological activity.

Based on this structure-activity relationship and assuming that 3 or 4 of the 2,3,7 or 8 positions are substituted with chlorine the number of potentially toxic isomers in each chlorine substitution group may be estimated as follows: 23 of the 75 PCDD and 40 of the 135 PCDF congeners are potentially toxic.

Using the criteria of chlorine substitutions in all 4 lateral positions and data from LD_{50} and enzyme induction studies, 5 PCDDs: 2,3,7,8-T_4CDD; 1,2,3,7,8-P_5CDD; 1,2,3,4,7,8-H_6CDD; 1,2,3,7,8,9-H_6CDD; and 1,2,3,6,7,8-H_6CDD; and 7 PCDFs: 2,3,7,8-T_4CDF; 1,2,3,7,8-P_5CDF; 2,3,4,7,8-P_5CDF; 1,2,3,6,7,8-H_6CDF; 1,2,3,7,8,9-H_6CDF; 1,2,3,4,7,8-H_6CDF; and 2,3,4,6,7,8-H_6CDF; can be identified as extremely toxic.

Toxicities or biological activities of PCDDs or PCDFs outside these two categories (extremely toxic and potentially toxic) drops rapidly with mono-, di-, and octa- substituted congeners showing little or no toxicity.

At present there is an absence of detailed analysis of the chemical and toxicological properties of the wide variety of PCDDs and PCDFs in environmental samples. However strong correlations appear between AHH induction, receptor binding avidity and toxicity with chlorine substitution patterns. This suggests the great potential of these tests (i.e. AHH induction and receptor binding avidity) for biological analysis (Section 6.2) and hazard assessment (Section 3.6.7) of environmental samples, especially mixtures.

2,3,7,8-T_4CDD Toxic Equivalents (Section 3.6.7)

Development of the relative toxicity factors of the various PCDD and PCDF isomers compared to 2,3,7,8-T_4CDD is based on a review of current research data. Chronic toxicity studies exist for only 4 PCDDs. Consequently the only practical approach to prorating chronic effects of PCDDs and PCDFs must be based on current knowledge of structural correlations derived from short term experiments.

PCDDs and PCDFs have similar toxicities, with the 2,3,7,8-substituted congeners in both groups being the most toxic. The 2,3,7,8-T_4CDD congener is the most toxic and most studied of the group and is used to derive the maximum allowable daily intake above. Other PCDDs and PCDFs are therefore prorated so that "2,3,7,8-T_4CDD toxic equivalents" can be derived.

It is recommended that a proposed numerical relationship for converting isomer group residue data to 2,3,7,8-T_4CDD equivalents be used. This relationship is based on the ratio of the acute toxicity or biological activity of the most toxic member of the PCDD or PCDF isomer group to that of 2,3,7,8-T_4CDD.

Estimation of Maximum Allowable Daily Intake (3.7)

Use of either the NOEL/Safety Factor or the Low Dose Extrapolation approach to risk estimation is arbitrary and very much dependent on the quality of the biological data available. It is the recommendation of the authors based on their scientific judgment of the evidence that the safety factor approach may be used when setting an adequate margin of safety to protect the citizens of Ontario.

From review of the literature, it is concluded that 2,3,7,8- T_4CDD produces tumours in rodents by an indirect mechanism. As such, it is concluded that 2,3,7,8- T_4CDD possesses a threshold level of dosage below which an increased rate of tumour production due to exposure to this compound would be unlikely.

The no-observed effect level (NOEL) identified in animal studies is used as an indication of where the threshold lies. A maximum allowable daily intake can be based on this level by applying a safety factor. In the case where long-term animal studies are available, it is generally agreed that a safety factor of 100 is appropriate (Nat. Acad. Sci., 1977).

This factor incorporates a number of considerations to account for uncertainty in extrapolating from animal data to humans, particularly an allowance in case humans are more sensitive than the animal species tested.

Since acute toxicity and long-term animal studies are available, and since the short-term mutagenicity studies, and the human epidemiology studies are generally negative, a safety factor of 100 is recommended.

The NOEL of 0.001 ug/kg/day for 2,3,7,8-T_4CDD, determined in the three-generation reproductive study of Murray et al. (1979) and the two-year oncology study of Kociba et al. (1978) both using rats, is recommended as a prudent basis for developing a maximum allowable daily intake for human PCDD and PCDF intake.

Thus using the NOEL of 0.001 ug/kg/day and a safety factor of 100 yields a maximum allowable daily intake of 1 x 10^{-5} ug/kg/day for humans. The dose from all sources for a 60 kg person would be 60 x 10^{-5} ug/day.

The recommended maximum allowable daily intake for total PCDDs and PCDFs is the equivalent of 10 pg 2,3,7,8-T_4CDD/kg body weight/day not to be exceeded on average over a year.

SOURCE/INPUT ASSESSMENT

Municipal Waste Incineration (Section 4.2.1.1)

Formation of PCDDs and PCDFs

On the basis of the extensive world literature dealing with the formation of PCDDs and PCDFs from the incineration of raw and treated municipal waste, it is apparent that the formation of these compounds is influenced primarily by the characteristics of the refuse burnt and the process operating conditions. In this regard, poor combustion conditions and the presence of PCDD/PCDF precursors or chlorine in the refuse can be expected to promote formation of PCDDs and PCDFs during combustion.

1. Most combustion theories implicate the formation of PCDDs and PCDFs either with the pyrolysis of chlorinated organic precursors already present in the refuse/sludge or to *de novo* synthesis involving chemically unrelated (non-chlorinated) organics and an inorganic chlorine source. The main support for the *de novo* formation hypothesis is that so far strong correlations between chlorinated organic precursors in the fuel and PCDD/PCDF formation have not been found. There are no positive monitoring data which demonstrate *de novo* formation; however, laboratory experiments show such mechanisms are possible.

2. Regardless of the formation mechanism(s) chlorobenzenes and chlorophenols are suspected of being major intermediate compounds.

3. On the basis of laboratory studies and considering the limitations involved in large scale combustion, it is generally apparent that proper operation of municipal incinerators within their design criteria together with careful control or regulation of operating parameters and collector devices represents the most attractive abatement strategy for reducing PCDD and PCDF emissions.

4. Although several attempts have been made to relate PCDD and PCDF emissions to various operating parameters or feed composition, the only correlation reported (Italian incinerator) has been higher emissions with minimal combustion temperature.

5. Ontario studies have shown that PCDDs/PCDFs are found in the feedstock to municipal incinerators. This should be an important consideration in future investigations to determine PCDD/PCDF mass balances or formation/destruction mechanisms.

Levels of PCDDs and PCDFs in Refuse/Precipitated Fly Ash/Stack Emissions

In an effort to correlate the large number of reports which have been published on the release of PCDDs and PCDFs from the incineration of raw and treated municipal waste the global data base has been tabularized and assessed for physical partitioning and isomer group distribution trends. References to total PCDDs or PCDFs means all residues analyzed in the tetrachlorinated (T_4CDD or T_4CDF) to octachlorinated (O_8CDD or O_8CDF) isomer groups. In isomer-specific studies, stack emissions and precipitated fly ash have been shown to contain about 30 PCDDs and 40 PCDFs. It must be emphasized that this type of analysis is subject to a number of caveats, the primary ones being great differences in incinerator design and operating conditions, composition of the refuse burnt, sampling techniques and the effect of various sampling artifacts on physical partitioning and isomer group distribution. As such the following summary statements should be considered as very approximate estimates of emission trends.

Refuse

PCDDs and PCDFs can enter municipal refuse or sewage sludge via herbicidal formulations, treated wood or PCB-containing products and other industrial or domestic wastes. Global data relating to this source of PCDDs and PCDFs are negligible. The only data available are from Ontario studies which indicate average concentrations from 0.5 - 19.8 ng total PCDDs/g and n.d. (not detectable) - 2.3 ng total PCDFs/g in municipal refuse. Data from Section 4.2.3.5 indicate that similar low ng/g levels of PCDDs and PCDFs are present in Ontario sewage sludge. These residues are mainly hepta- and octachlorinated PCDDs and PCDFs.

Precipitated Fly Ash

Incinerator ash is disposed of in secure landfill sites (Section 4.2.3.4).

1. On a global basis (including Ontario) total PCDDs and PCDFs appear to be equally distributed in precipitated fly ash averaging about 1 ug/g each; however, Ontario studies indicate total PCDD concentrations in precipitated fly

ash are about fifteen times lower, averaging about 80 ng/g each. In two of the three Ontario studies where both PCDDs and PCDFs in fly ash were analyzed, the total PCDF level ranged up to two times the corresponding total PCDD concentration.

2. On a global basis, the T_4CDD isomer group only represents about 3% of the total PCDDs and PCDFs in precipitated fly ash. This percentage increases through the PCDD series to about 14% for O_8CDD. In contrast the T_4CDF isomer group comprises about 13% of the total PCDD and PCDF content of fly ash and as the chlorine substitution of PCDFs increases the percentage contribution drops to about 2% for O_8CDF.

3. The major T_4CDD isomers are 1,3,6,8-T_4CDD and 1,3,7,9-T_4CDD present at about 20% and 12%, respectively, of the total T_4CDD. The most toxic isomer (2,3,7,8-T_4CDD) comprises about 5% of the total T_4CDD and 0.2% of the total PCDD and PCDF.

Stack Emissions

Total stack emissions consist of a combination of stack collected particulate material and gaseous or aerosol (gas phase) components which escape collection by pollution abatement equipment. In-stack sampling can give rise to several artefacts at the collection point and the physical partitioning of the escaping contaminants by the sampling device into particulate and gaseous phases may not accurately depict the resulting concentrations in air following atmospheric dispersion.

Subject to the foregoing caveats the following summary points have been prepared from the analysis of the global data base.

1. On a weight:weight basis (ng/g) total PCDDs and PCDFs in stack-collected particulates (escaping the pollution abatement devices) are about twice to seven times as high as corresponding results for precipitated fly ash.

2. On the basis of all studies in which total stack emissions (particulate plus gaseous) have been reported, the average PCDD and PCDF concentrations are both 4 ug/m^3. However, when only those studies which reported results for total (T_4 - O_8) PCDDs and PCDFs were considered, PCDDs comprised only 35% of total PCDDs plus PCDF emissions.

3. On a global basis, about 5% of the total PCDD plus PCDF in total stack emissions consists of T_4CDD; while T_4CDF comprises about 30% of total PCDD plus PCDF stack emissions. In the four Ontario studies in which total stack emissions have been presented as chlorinated isomer groups, the proportion of T_4CDD and T_4CDF in total PCDD plus PCDF stack emissions was similar to the global trend with average values of 8 and 26% respectively.

4. In all studies (global data base) in which both particulate and gaseous results have been reported, total PCDD and PCDF gaseous concentrations comprised about 75% of the total stack emissions. In the 1984 Ontario study the gaseous and particulate partitioning was about 50:50.

5. The composition of major T_4CDD isomers in stack emissions is similar to that described for precipitated fly ash.

Chemical Waste Incineration (Section 4.2.1.2)

The destruction of chemical wastes, e.g. PCB wastes, is a viable procedure when conducted under proper conditions. (i.e. high combustion temperatures and sufficient residence times) in appropriate incineration facilities.

On the basis of actual industrial and experimental incinerator facilities and micro-scale pyrolysis experiments, it can be concluded that:

1. Under some conditions the potential exists for PCDD and PCDF formation and emission into the atmosphere during the incineration of waste chemicals.

2. With proper incinerator design and operation, chemical wastes can be effectively destroyed with negligible emissions of PCDD/PCDF to the atmosphere.

Accordingly, each site (incinerator) must be examined on the merits of the operational parameters before any general PCDD/PCDF emission factors can be estimated.

Biological Waste Incineration (Section 4.2.1.3)

Large amounts of pathological wastes including human or animal tissues, plastics, chlorinated disinfectants and drugs from hospitals or other medical or veterinary institutions are disposed of on-site in heating boilers. Cremation of human or animal tissues at cemeteries might also be considered under this category. No definite conclusion concerning the potential of these practices for PCDD or PCDF emissions can be made since no on-site Ontario investigations have been reported yet. The only confirmation consists of emission data from an incinerator at a B.C. hospital.

Wood Combustion (Section 4.2.1.4)

A. Incineration of Chemically Treated Wood Products

The former commercial use of chlorophenols in paper mills as fungicides and slimicides and the continuing use of these compounds as wood preservatives results in incineration of treated wood waste e.g. pulp, sawdust and shavings and old treated wood products such as wood packaging and railroad ties.

Laboratory and pilot-scale pyrolysis studies show that ug/g amounts of PCDDs and PCDFs can be released in combustion gases and emitted particulates. It is clear that the burning of wood and wood products contaminated with chlorophenols represents a significant potential source of PCDDs and PCDFs into the atmosphere. This situation is complicated by the fact that:

1. Much of the burning of this type of material is conducted in 'open fire' situations where control of temperature and residence time is not possible, and,

2. The amount of chlorophenol treated wood/wood products burned in Ontario each year is unknown but is currently under investigation by Environment Canada.

B. Combustion of Untreated Wood

The growing use of wood as a residential heating fuel has led to several studies of PCDDs and PCDFs in soot and flue gas particulates from fireplaces and wood stoves.

Many questions still remain unanswered in regard to the formation and release of PCDDs and PCDFs into the atmosphere (wood type, furnace construction, operator bias). Additional quantitative information is required in order to establish a reliable estimate of the magnitude of the emissions.

It can, therefore, be concluded only that the combustion of wood for heating or aesthetic purposes does represent a potential source for the release of PCDDs and PCDFs into the atmosphere.

Additional research including analysis of emissions from wood burning devices as well as suspended particulates in the atmosphere during winter months is required to quantify this component of the total atmospheric loading of PCDDs and PCDFs.

Fossil Fuel Combustion (Section 4.2.1.5)

Large quantities of fossil fuels e.g. coal and oil are used for power generation and central heating in this province. Most studies of such facilities have been limited to studies of PCDD and PCDF levels in fly ash, though two studies of flue gases and stack emissions from coal-fired sources have been reported.

On the basis of this information, it can be concluded that only coal-fired power plants have demonstrated any significant potential as a source of atmospheric PCDDs and PCDFs. In view of the limited data base investigation of coal and oil-fired power plants as sources of PCDDs and PCDFs should continue.

Other Combustion Sources (Section 4.2.1.6)

A. Electrical Capacitor/Transformer Fires

Analysis of the combustion products of fires or explosions involving PCB- or PCB and chlorinated benzene-filled transformers and capacitors in Sweden, the U.S. and Canada indicates the formation of ug/g levels of PCDFs and/or PCDDs. The accidental combustion of transformer oils can be considered a direct input of PCDFs and/or PCDDs into the environment. While only limited dispersion of these contaminants occurs, direct human exposure will be averted if PCB- and chlorinated benzene containing equipment is isolated in large or public buildings.

B. Forest Fires

Limited studies of the experimental combustion of herbicide-treated vegetation and the work on untreated wood combustion (Section 4.2.1.4) suggests that natural fires represent a potential source of PCDDs and PCDFs. Analysis of atmospheric smoke emissions during actual fires, residual ashes and sediment core analysis of neighbouring bodies of waters of PCDDs and PCDFs is required to confirm this speculation.

C. Automotive Exhaust and Cigarette Smoke

These activities are potential but unconfirmed sources of PCDDs and PCDFs.

Estimates of Atmospheric Input to the Ontario Environment from Combustion Sources

(Section 4.2.1.7)

A. Municipal Incinerators

Based on stack emission tests for PCDDs and PCDFs conducted on two main municipal refuse incinerators (SWARU, Hamilton and Commissioner's Street, Toronto) and the main Ontario sewage sludge incinerator (Ashbridges Bay, Toronto) and their daily operational status, the total PCDD and PCDF emissions for these three Ontario sources is approximately 8 and 14 kg/per year respectively for total PCDD plus PCDF emission of 22 kg per year.

B. Wood

Ontario's forest industry uses an estimated 2 billion kg of wood wastes annually in various energy recovery facilities. About 5 billion kg of wood are burned annually as residential heating fuel. Forest fires and incineration of chlorophenol-treated railway ties, shipping containers and other treated wood waste also represent unknown but potential PCDD and PCDF inputs into the Ontario environment.

C. Coal

Ontario uses about 18 million tonnes of coal annually. It is estimated that the total annual PCDD and PCDF output from this would be about 50 and 470 g, respectively.

D. Pathological Waste

Ontario hospitals incinerate millions of kilograms of this type of waste every year. Although tests have confirmed the emission of PCDDs and PCDFs from a hospital incinerator in B.C. no such tests have been performed in Ontario. This potential source of PCDDs and PCDFs requires further investigation in Ontario before estimates of input can be attempted.

E. Chemical Waste, Electrical Capacitors, Cigarettes, Automotive Emissions

No information which would assist in quantifying the possible emission of PCDD/PCDF from these source types in Ontario is available.

Summary

In view of the limited amount of information concerning the potential formation and release of PCDDs and PCDFs into the Ontario environment from the combustion of other materials including wood, coal, pathological waste, chemical waste, electrical capacitors, cigarettes and automotive fuels, it is not possible to derive any firm PCDD/PCDF output estimates. However, the information which is available concerning the combustion of these materials in Ontario has been presented in order to provide some relevance to their possible significance at a later date.

CHEMICAL MANUFACTURING AND USE (Section 4.2.2)

Chlorophenol Production and Use (Section 4.2.2.1)

Chlorophenols are not produced in Ontario. Dichlorophenols (D_2CP) and trichlorophenols (T_3CP) are mainly used as intermediates in the manufacture of pesticides whereas tetrachlorophenols (T_4CP) and pentachlorophenol (P_5CP) are used directly as fungicides.

P_5CP, 2,3,4,6-T_4CP and their sodium salts are mainly used as wood preservatives. It is estimated that Ontario uses about 800 tonnes of these chemicals annually.

There are two major users of P_5CP and 2,3,4,6-T_4CP for treated wood products. The other main use of P_5CP is for the in situ treatment of wooden public utility poles.

P_5CP is contaminated with mg/kg levels of PCDDs and PCDFs mainly the hexachlorinated, heptachlorinated and octachlorinated forms. Based on the current formulation in use, the annual PCDD and PCDF input to the Ontario Environment from this source is estimated at about 503 kg.

Impact of PCDDs and PCDFs on the environment resulting from the use of chlorinated phenols will involve a) localized inputs to soil and water near wood preservation facilities and retreated utility poles, and b) more widespread soil and water inputs involving distribution and use of treated wood products. Direct inputs to the atmospheric environment are expected to be minimal.

Agriculture Canada and U.S. Environmental Protection Agency have proposed restrictions on the application, use and PCDD content of P_5CP and its salts which should substantially reduce the PCDD and PCDF input from this source in the future.

Herbicide Production/Formulation/Use (Section 4.2.2.2)

No phenoxy herbicides are produced in Ontario at present. 2,4-dichlorophenoxyacetic acid (2,4-D) and its various ester and amine formulations are selective herbicides used for the control of broadleaved weeds on grain crops, hydro rights-of-way, residential lawns and public parks. It has been shown that they contain lower chlorinated PCDDs, e.g., 2,7-D_2CDD; 1,3,7-T_3CDD; 1,3,6,8-T_4CDD and 1,3,6,9-T_4CDD. Current formulations contain less than 2 ug of any PCDD congener/kg of technical product. Other phenoxy herbicides used in Ontario which may contain PCDDs are 2,4-DP, MCPA, MCPB and Dicamba. 2,4,5-T which contained 2,3,7,8-T_4CDD has not been applied in Ontario since 1980. The total annual amount of PCDDs estimated to enter the Ontario environment as a result of phenoxy herbicide usage is less than 12 g.

PCB Manufacture and Use (Section 4.2.2.4)

Polychlorinated biphenyls (PCBs) have been widely used as dielectric fluids and heat transfer agents in electrical transformers and capacitors since 1930. PCBs were also used in a wide variety of common products.

PCDFs have been detected in low ug/g amounts in most PCB preparations. Fresh PCBs do not contain PCDDs, however low levels of PCDDs and elevated levels of PCDFs may form in PCB formulations following prolonged heating or fires especially if chlorinated benzenes are also present in the formulation.

Production of Aroclors (North American PCBs) ceased in 1976.

Over 90% of all PCBs still in use in Canada are in Ontario. Current estimates of PCB quantities in Ontario are approximately 6.5 million litres or 9.75 million kg still in use in transformers and capacitors. Another 1.5 million litres or 2.25 million kg PCBs are in storage in 18 major centres and 160 smaller sites around the province awaiting disposal.

Aroclors contain 1 to 2 ug total PCDFs/g PCB; therefore, combined PCBs in use or storage around the province may contain 12 to 24 kg PCDFs. However these PCDFs are in closed containers and do not represent a direct input to the environment.

Therefore, input of PCDF to the environment from use of Aroclor would be through:

i) old transformer leakage;
ii) fires involving transformers containing Aroclors (discussed above under "Other Combustion Sources");
iii) disposal in waste dumps of old transformer/dielectric fluids containing Aroclors.

These amounts cannot be estimated from available information. No data for Ontario are available from which estimates of PCDF input to the environment from i) or iii) can be made.

Commercial/Domestic Products/Use (Section 4.2.2.5)

Input of PCDDs and PCDFs in domestic products such as paints, stains, wood preservatives, health care products, disinfectants, weed control and lawn care products cannot be estimated from the data reviewed. However, while the quantities involved may be low they will result in direct human exposure.

Waste Disposal Sites (Section 4.2.3)

Disposal sites where chemicals or chemical waste have been dumped can be considered as potential sources of PCDDs and PCDFs into the environment. Sites where soil or water contamination may occur are described below.

Elmira (Section 4.2.3.1)

Since 1941 the Uniroyal Chemical division of Uniroyal Ltd. in Elmira has produced a wide range of organic chemicals for the agricultural, rubber and plastics industries. From 1950 to 1969, Uniroyal produced the herbicide 2,4,5-T.

Extensive testing of 11 private wells, 3 municipal production wells, 3 municipal test wells, 2 containment wells on Uniroyal property, 42 test wells on Uniroyal property and 6 surface water locations have shown that only one shallow test well, TW18S contains consistently reproducible levels of PCDDs and PCDFs. Repeat samples of other test wells were negative for PCDDs and PCDFs at the 0.02 (T_4CDDs) to 0.3 (O_8CDD) ng/L level of detection.

No PCDDs or PCDFs have been identified in surface water or fish in the area. Presence of 2,3,7,8-T_4CDD in the T_4CDD fraction from TW18S has not been confirmed. Wastes from the production of 2,4,5-T are contained in drums buried on the site. The concentration of 2,3,7,8-TCDD in this waste is not known with any degree of accuracy; therefore, any estimate of total amounts is speculative.

Wood Preservation Sites (Section 4.2.3.2)

The two major users of P_5CP in wood preservation in Ontario are Abitibi Price/Northern Wood Preservers, Thunder Bay and Domtar Wood Preservers, Trenton. Major sources of PCDD and PCDF input to the environment may occur via disposal of P_5CP-contaminated process waters, P_5CP sludges from pressure treatment cylinders or wood dipping tanks, or P_5CP-contaminated liquids, soils or treated wood waste.

Effluents from Ontario wood preserving plants have not been analyzed for levels of PCDD/PCDF. Therefore, no estimate of PCDD or PCDF environmental input from this source can be made.

Landfill Sites Accepting Chemicals and Other Industrial Sites Potentially Contaminated with PCDDs or PCDFs (Section 4.2.3.3)

A preliminary list of sites potentially contaminated following the production/use/disposal of wastes from chlorophenol products, phenoxy herbicides or PCBs has been prepared. Generally no records or data are available on quantities or types of materials historically deposited at these sites.

Incinerator Ash Disposal Sites (Section 4.2..3.4)

Fly ash produced from incineration of municipal garbage contains levels of PCDD plus PCDF from 0.03 ug to 6.2 ug/g. However, concentrations of PCDDs and PCDFs in bottom ash from Ontario incinerators are much lower ranging from ND to 0.007 ug/g (limit of detection - 0.2 to 1.0ng/g).

Estimates of PCDD and PCDF loading from incinerator ash to landfill sites for two Ontario municipal incinerators, SWARU, Hamilton and Commissioner Street, Toronto and the main Toronto sewage sludge incinerator at Ashbridges Bay are less than 1 kg/year. No data exist on the degree of mobility of PCDDs or PCDFs from ashes in a landfill site.

Municipal/Industrial Sewage and Sludge Disposal (Section 4.2.3.5)

Ontario produces from 150,000 to 300,000 dry tonnes of municipal and industrial sewage material annually. About 50% is incinerated, and the rest is landfilled, applied to farm land or drying beds, used for composting or used in mine tailing reclamation projects. Very limited investigations of municipal sewage sludges from Ontario indicate that ng/g levels of PCDDs and PCDFs are present. Further studies of these low levels of PCDDs and PCDFs in sewage sludge are required before reliable estimates of PCDD and PCDF input to the Ontario environment can be made.

TRANSBOUNDARY SOURCES

Niagara Frontier (Section 4.2.4.1)

Two hundred and fifteen waste disposal sites have been identified along the Niagara River. The Love Canal and Hyde Park disposal sites have definitely received large quantities of hazardous waste. Estimates of the total amount of PCDDs and PCDFs buried in these sites vary from 100 kg to 5000 kg. 2,3,7,8-T_4CDD has been identified in water, sediments and fish from the Niagara River suggesting migration of PCDDs and PCDFs from the waste sites to the river and ultimately into Lake Ontario. It has been estimated that 2 to 20 kg of PCDDs plus PCDFs flow annually into Lake Ontario.

Dow, Midland, Michigan (Section 4.2.4.2)

Effluents from Dow Chemical, Midland, Michigan as well as fish from the Tittabawassee River near Dow have been found to contain levels of PCDD and PCDF. However, data from fish caught in Lake Huron indicate no detectable 2,3,7,8-T_4CDD (detection 10 ppt) for whitefish, lake trout, rainbow trout and walleye.

Therefore, the contribution of Dow Chemical effluents to the Ontario Environment can be assumed to be minimal.

St. Clair River/Detroit River (Section 4.2.4.3)

Gull eggs from Fighting Island, Detroit River have been found to contain levels of 40 ng/kg of 2,3,7,8-T_4CDD.

Also, low levels (80-90 pg/g) of H_7CDD and O_8CDD have been detected in two core samples of soils from Fighting Island (Windsor Star, 1984)

Walleye caught in Lake St. Clair in 1979 showed no PCDD levels. However, low pg/g levels of PCDFs were found in the same fish.

From these data, it appears that the St. Clair and Detroit River areas may have some contamination by PCDDs and PCDFs, the magnitude of which cannot be assessed at this time.

Ontario Water Utilities (Section 4.2.4.4)

Most Ontario water treatment plants obtain their raw water supply from the Great Lakes, an international water supply. Intensive sampling of raw and treated water from widely distributed treatment plants has shown that PCDDs and PCDFs are not detected in Ontario drinking water even using an analytical methodology sensitive to 0.005 ng/L. "Ultratrace" amounts of PCDDs in the 0.01 to 0.05 ng/L range were detected in samples of raw water from the western end of Lake Ontario. Clearly conventional water treatment procedures are effective in removing the trace amounts of PCDDs or PCDFs occasionally present in raw water.

ENVIRONMENTAL TRANSPORT, DISTRIBUTION AND FATE ASSESSMENT

Atmospheric Transport (Section 4.3.1)

Although the mechanisms affecting stability and transport of PCDDs in the atmosphere are not fully known there is a growing base of environmental monitoring data which should prove useful in evaluating these concepts. No attempt has been made to consolidate these results due to their limited number and to the wide range in the atmospheric component measured (dust fall, suspended particulates, gaseous).

Deposition, Fate and Mobility in the Terrestrial Environment (Section 4.3.2)

Deposition to Soils and Plants from Atmospheric Sources (Section 4.3.2.1)

The deposition of PCDDs to soil surfaces has been the subject of extensive investigations following the accidental release of 2,3,7,8-T_4CDD at Seveso, Italy; the disposal of still bottom contaminated waste oils for dust control purposes in Missouri; and the testing and storage of T_4CDD contaminated herbicides which were used by the U.S. Air Force for war-related defoliation practices in South-East Asia.

In Ontario, a soil sampling survey was initiated in 1983 in the vicinity of a major municipal refuse incinerator in Hamilton to determine if PCDDs and PCDFs identified in stack emissions had accumulated in surface (0-5cm) soils. No concentration gradients for any of the PCDDs or PCDFs were apparent when compared with distance or direction relative to the incinerator or with their proximity to the calculated maximum ground level concentration (1182m).

All 14 of the soil samples had detectable quantities of at least one of the five PCDD (tetrachlorinated to octachlorinated) isomer groups whereas only eight samples contained detectable levels of one or more PCDF congeners. Only one site had a measurable quantity (0.007 ng/g) of T_4CDD in the soil. The highest PCDD concentration was 3.5 ng/g O_8CDD; however, a similar level of 3.2 ng/g O_8CDD was found in the soil at one of the urban control sites well remote from the plant. The soil at the remote/rural control site contained 0.810 ng/g O_8CDD. The levels of the other PCDD and PCDF congeners in soil from the 11 sites around the plant generally were within the range detected in the urban control soils.

In addition to direct contamination from atmospheric emissions, accidental spills and the use of contaminated phenoxy and chlorophenol products, are the other major possible sources of PCDDs and PCDFs in soils.

Degradation in Soils and Plants (Section 4.3.2.3)

Photodegradation

PCDDs and PCDFs readily undergo photolysis by ultraviolet light in the presence of hydrogen donors including alcohols, ethers, hydrocarbons, and natural products such as waxes.

Considering all the natural removal mechanisms and the fact that soil surfaces and plant foliage are the major direct recipients of pesticide sprays and combustion products, photolysis appears to be the most significant degradation process; however, quantitatively little is known concerning the rate of photolysis in natural systems.

Biodegradation

T_4CDD has been found to be a relatively stable compound capable of resisting microbial metabolism and thus can be expected to remain in the soil for long periods of time. However, on the basis of recent controlled model ecosystem studies, using carbon labelling techniques, there is evidence for very slow metabolism by some micro-organisms. As a result of the resistance of T_4CDD to microbial activity, PCDDs are considered to be highly persistent with a reported half-life in soils in excess of 10 years. Unfortunately, no information has been generated for any of the other PCDD and PCDF congeners.

Physical Transport in Soils (Section 4.3.2.4)

The movement of PCDDs in soil over time is complicated by the effect of other factors which influence its fate in soil systems, i.e., microbial degradation, photodegradation, and volatilization. Thus, although several studies have been conducted which address the movement through the soil profile, there is as yet no general consensus on the mechanism involved.

Aquatic Fate and Persistence (Section 4.3.3)

Dissipation of PCDD and PCDF residues in the aquatic environment is influenced by non-biological and biological processes. Non-biological processes include volatilization, and photolysis. Biological processes include microbial decomposition and to a lesser extent metabolism in other aquatic organisms. Environmental studies of PCDDs and PCDFs in laboratory model ecosystems, pond studies and studies of contaminated areas indicate that PCDDs and PCDFs are more frequently found at measurable concentrations in sediment than in water. Sediment core analysis also indicates that PCDDs and PCDFs may persist in lake sediments for decades.

Biological Accumulation/Magnification
(Section 4.3.3)

Wildlife and Domestic Animals (Section 4.3.4.1)

The uptake, absorption, distribution and excretion of PCDDs and PCDFs by laboratory animals is considered in Sections 5.2.3 and 5.2.4.

On the basis of studies of wild and domestic animals at Seveso or in areas where the herbicide 2,4,5-T has been sprayed it can be concluded that bioconcentration factors for soil to mammals and food to mammals are low.

Limited analysis of Ontario poultry, pork, beef and eggs has shown that residues of some PCDDs and PCDFs are present in a percentage of the samples analyzed.

It is recommended that MOE monitor other food items (in conjunction with Health and Welfare Canada) to quantify contributions from these sources.

Vegetation (Section 4.3.3.2)

Studies of plants at Seveso and areas where 2,4,5-T was sprayed indicate that plants do not bioconcentrate T_4CDD from the soil. This is an important consequence since plants are the basis of the food web.

Aquatic Biota (Section 4.3.4.3)

Laboratory and field studies indicate that fish bioaccumulate PCDDs and PCDFs by 10^2-fold to 10^3-fold factors. Considering the hydrophobic nature of PCDDs and PCDFs this bioaccumulation is quite moderate. Limited evidence from laboratory and field studies suggests that T_4CDD and T_4CDFs are preferentially accumulated.

Residue Levels in Fish From Ontario Waters
(Section 4.3.4.4)

The Ministry of the Environment Dioxin Laboratory has analyzed many fish samples from the Great Lakes, interconnecting waterways and some tributary streams. Except for trace amounts of 2,3,7,8-T_4CDD in lake trout from Peninsula Harbour, the fish of Lakes Superior, Huron and Erie appear to be free of this contaminant. In contrast, 2,3,7,8-T_4CDD is often detected in samples from the Niagara River and Lake Ontario.

Large lake trout from Western Lake Ontario represent a source of human exposure as indicated in the "Guide to Eating Ontario Sportfish".

Recent data and studies discussed in Section 4.2.4.3 show that some Great Lakes fish which do not contain 2,3,7,8-T_4CDD may have high pg/g levels of other PCDDs and PCDFs. These data are area-specific. Further monitoring of PCDD and PCDF levels in Ontario fish is required. Future sport fish consumption guidelines should be updated in light of these data.

Residue Levels of PCDDs and PCDFs in Humans

(Section 4.3.4.5)

Human exposure to PCDDs and PCDFs may occur occupationally, accidentally, or from dietary intake and/or environmental exposure e.g. air, water or soil.

PCDFs were not detected (detection limit - 10 ng/kg) in the blood of Japanese workers handling fresh or used PCB formulations. Similarly, no PCDFs were detected in the blood of healthy Japanese not exposed to PCBs or contaminated rice oil.

Mean 2,3,7,8-T_4CDD levels of 5 to 12 ng/kg were found in adipose tissue of people from the Kingston and Ottawa areas, a U.S. urban population and U.S. veterans not exposed to Agent Orange herbicide.

Recently reported surveys of PCDDs and PCDFs in North American adipose tissue indicate that total

PCDD plus PCDF residue levels range from 700 to 1700 ng/kg. Characteristic profiles of about 12 PCDDs and PCDFs substituted in the 2,3,7, and 8 positions were found. PCDFs represent about 10% of the total residues found. Levels of the toxic 2,3,7,8-T_4CDD and 2,3,4,7,8-P_5CDF congeners represent only 1 - 2% of the total residues. By contrast, the relatively non-toxic O_8CDD congener formed 75 - 80% of these total residues.

EXPOSURE ASSESSMENT

The purpose of this exposure assessment is:

(i) to estimate current health risks in comparison with data on toxic effects;

(ii) to indicate the relative proportion of the daily dose from various sources to aid in setting standards for separate environmental media;

(iii) to give examples of how the toxic equivalents approach would be used in actual practice;

(iv) to compare routes of exposure with measured body burdens to determine whether all sources or routes of exposure have been taken into account;

(v) to identify potential problem areas for future investigation.

It is important to consider that the exposure assessments presented are hypothetical and should not be used to infer or come to conclusions about health risks to specific individuals or in specific geographical areas.

Exposure Pathways (Section 5.2)

Maximum estimated concentrations from the various sources and the 2,3,7,8-T_4CDD toxic equivalents are calculated as follows:

1) Ambient Air (Section 5.2.1)	- estimates of maximum annual average ground level concentrations range from 2 to 27 pg/m^3 of PCDDs and from 3 to 70 pg/m^3 of PCDFs in the vicinity of Ontario municipal waste and sludge incinerators - the calculated worst case 2,3,7,8-T_4CDD equivalent concentration for ambient air in this exposure assessment is 8.4 pg/m^3 on an annual basis
2) Water (Section 5.2.2)	
Surface Water	- maximum ambient concentrations range from ND to 46 pg/L for PCDDs - the calculated worst case 2,3,7,8-T_4CDD equivalent ambient concentration for use in this exposure assessment is 0.002 ng/L
Drinking Water	- none (detection limit 0.005 - 0.01 ng/L).
3) Soil (Section 5.2.3)	- maximum measured soil concentration near incinerators is 4640 pg/g

of PCDDs and 180 pg/g of PCDFs

- the calculated 2,3,7,8-T_4CDD equivalent soil concentration near incinerators is 81.1 pg/g of PCDDs plus PCDFs.

4) Food
(Section 5.2.4)

a) Fish
- concentrations in fish range from non-detectable to 60 pg/g of 2,3,7,8-T_4CDD.
- only T_4- and P_5CDDs and T_4- and P_5CDFs are detected
- note the maximum allowable concentration of 2,3,7,8-T_4CDD in sportfish for consumption in Ontario is 20 pg/g.

b) Other Food
- significant pg/g quantities of H_6-, H_7-, or O_8CDDs have been detected in pork, poultry and eggs
- maximum 2,3,7,8-T_4CDD toxic equivalent concentrations are 0.01 pg/g (eggs), 0.8 pg/g (pork fat) and 8.8 pg/g (chicken fat).

Estimated Doses (Section 5.3)

Ranges of possible daily doses of 2,3,7,8-T_4CDD or its toxic equivalent of PCDDs and PCDFs have been estimated. It is unlikely that all of the maximum doses would be received by any one individual or group of individuals due to the variety of site-specific exposures which give rise to the estimated maximums.

Exposure Scenarios, Special Groups at Risk and Current Levels of Exposure (Section 5.4)

Compensating for the variable toxicity of the mixtures of PCDDs and PCDFs to which the population of Ontario may be exposed by using 2,3,7,8-T_4CDD toxic equivalents, various exposure scenarios have been estimated. From this analysis special groups at risk can be identified: people living near some, but not all combustion sources, who inhale PCDDs and PCDFs adsorbed on airborne particulates, people eating contaminated fish or other dietary items and children ingesting urban soils.

Only the contaminated fish contains a high proportion of 2,3,7,8-T_4CDD in the total PCDDs/PCDFs measured. In the other scenarios, comparison of the calculated dose from total PCDDs and PCDFs with the recommended maximum daily dose based on 2,3,7,8-T_4CDD results in an overestimation of the health risk.

Information from source assessment (Section 4.2.5) and two Federal reviews of pathways of human exposure indicate that occupational groups involved in the formulation or use of chemical products containing PCDDs or PCDFs are probably the most highly exposed.

The current level of exposure of the population of Ontario to 2,3,7,8-T_4CDD or its toxic equivalent from all sources, based on analysis of residues in human adipose tissue, is estimated to be 7.2 pg toxic equivalent/kg body weight/day.

Implications for Risk Management (Section 5.2.8)

The following conclusions regarding the estimated exposure assessment and the recommended maximum allowable daily intake derived in this document, may be useful to decision makers involved in setting the standard for the various environmental media and in risk management:

1. Exposure is not consistent over a lifetime. For example:

    (a) People and children would not be exposed to contaminated soil on a continuous daily basis (i.e. during snow cover and inclement weather).

    (b) Indoor particulate and soil levels may be 10 to 100- fold lower than outdoor levels. Considerable time is spent indoors.

    (c) Contaminated surface water, elevated levels of PCDDs in fish and some food items are very low probability situations.

    (d) People in the vicinity of combustion sources are not continually exposed to maximum ground level concentrations of emissions.

2. Estimates of the current levels of exposure indicate that these levels are well below the recommended maximum allowable daily intake.

3. Bioanalytical methods should provide simple, cost-effective monitoring and early warning tools for present and potential future problems.

4. Development of the environmental standards requires consideration of:

   (a) the recommended maximum allowable daily intake of PCDDs + PCDFs equivalent to 10 pg 2,3,7,8-T_4CDD/Kg.bw./day;

   (b) establishment of protocols for methods of analytical measurement in the various matrices (i.e., air, soil, water, etc.), see Sections 6.1 and 6.2;

   (c) the development of standards for allowable levels in the various matrices (i.e., air, soil, water);

   (d) development of monitoring programs of emissions from stacks, waste sites, waste waters, etc.

5. Possible approaches for development of standards for the various matrices are:

   (a) Apportion the maximum allowable daily intake of 10 pg 2,3,7,8-T_4CDD equivalent/kg.b.w./day equally among the various media: soil, water, food, air;

   (b) Indicate different maxima for the various media so that no one route of exposure can be allowed the total maximum allowable daily intake;

(c) Allocate the maximum allowable daily intake to one route of exposure;

(d) Assume water accounts for 10% of the maximum allowable daily intake;

(e) Use current limits of detection to apply more stringent levels of control;

(f) Not to measurably increase the PCDD and PCDF content of an environmental matrix above current background levels;

(g) Use guidelines proposed by other regulatory agencies.

6. Combinations of the approaches discussed in point 5 above might be used. For example, option (e) could be applied to drinking water and option (f) to air, soil and diet (fish consumption).

7. Another consideration in apportioning the maximum allowable daily intake is to consider special groups at risk. For example, occupationally exposed persons as compared with other members of the population who have little or no exposure to PCDDs or PCDFs.

ANALYTICAL METHODS FOR MONITORING PCDDs and PCDFs

PCDDs and PCDFs: Review of Analytical Methods (Section 6.1)

Sampling methods, sample extraction, sample cleanup, and the most effective instrumental techniques were reviewed. Quality control procedures, quantification methods and limitations of existing methods were examined. Recent reports of the analysis of PCDDs and PCDFs at parts-per-quadrillion concentrations in water were achieved by high concentration factors, not by improvements in the basic analytical methodology.

Biological Analysis (Section 6.2)

Several methods of analysing various media for PCDDs and PCDFs using biological techniques are currently under development. These areas of biological analysis are: radioimmunoassay, AHH induction, cytosol receptor assay, tissue cultures and whole animal tests. Each of these tests is based on the biological/biochemical properties of PCDDs and PCDFs. Current limitations include limited experience and validation and lack of specificity and sensitivity (when compared with current analytical techniques). Potential advantages of these tests include low cost, large-scale screening ability and biological realism.

The bioassays that show the most promise as analytical tools to measure the "2,3,7,8-T_4CDD equivalent" activity of environmental extracts are the AHH induction and cytosol-receptor assays.

Conclusions and Recommendations on Current Limitations and Future Research Needs (Section 6.3)

Limitations to Existing Data Base

1. Most studies of incinerators are not comprehensive enough to make useful correlations between PCDD/PCDF emissions and other factors. i.e. either fly ash or stack samples taken, either PCDD only or 2,3,7,8-T_4CDD only measured. Few studies present analysis of PCDD/PCDF as well as possible precursors.

2. Only Ontario studies have included analysis of feedstock as well as fly ash/stack emissions. This total analysis is needed to develop PCDD/PCDF mass balances in incinerators.

3. Most investigators in the field of incinerator emissions admit the difficulties in comparing data from different studies. Few have stated, however, that because of the great differences in incinerator design, feedstock composition, and operating conditions of municipal incinerators, there is no *a priori* reason to expect emissions from these sources to be comparable. Each incinerator must be evaluated separately to accurately assess emissions.

4. Seasonal variations in incinerator emissions have not been thoroughly evaluated.

5. Reproducibility studies for stack emission quantitative determination of PCDD/PCDF have not been performed.

6. More analytical standards, especially of the more toxic congeners, are needed. Many have now been synthesized in private or research laboratories, but are still not available from commercial sources. Use of standards is often not discussed in studies that present quantitative data.

7. Much of the work to date has ignored the PCDFs.

8. It is not clear in most cases how the reported emissions or concentrations of PCDD/PCDF relate to human health, especially in terms of bioavailability.

9. Many studies report data without giving key experimental details such as recoveries, cleanup methods, confirmation techniques used, *etc*.

10. Background levels are generally unknown; these should be determined since it is now apparent that low level PCDD/PCDF contamination is much more ubiquitous than previously suspected.

11. Precision and accuracy of analytical data are often not specified or even determined. Many more round-robin laboratory studies and standard exchanges are needed.

12. Information on the biological and toxicological properties of PCDDs and PCDFs other than 2,3,7,8-T_4CDD is extremely limited.

13. Despite studies of occupational exposure and investigations of environmental exposure, accurate assessment of human exposure to PCDDs and PCDFs for the purposes of estimating human health effects has not been achieved. The validity of extrapolating these effects from laboratory animal studies is questionable.

Recommendations on Future Research Required

1. All 210 PCDD/PCDF compounds should be available individually from commercial laboratories in crystalline form or in certified correct concentration solution.

2. Additional ^{13}C-labelled standards are needed.

3. Increased quality control is needed for quantitative analysis.

4. Round-robin studies for samples such as incinerator fly ash are needed.

5. Reproducibility of the entire analytical process including sampling must be determined for incinerator emissions testing.

6. Future incinerator testing should include feed stock analysis.

7. Background levels of PCDD/PCDF in various environmental media in Ontario, especially urban air should be determined.

8. Some work of Dow Chemical should be confirmed by others i.e. PCDD/PCDF formation in

cigarettes and automobile exhaust. Further studies of PCDD/PCDF emissions from woodburning (especially from forest fires) and fossil fuel burning should be conducted.

9. Extensive toxicity testing of PCDDs and PCDFs suspected of being toxic based on structure-activity predictions is required. Such testing should include short term acute toxicity studies, AHH induction assays, *in vitro* and *in vivo* genotoxicity testing using well-validated microbial and mammalian test systems, as well as, studies of reproductive function and long term carcinogenicity bioassays. Some attempt should be made to investigate the effects of mixtures of PCDDs and PCDFs.

10. Methods of determining the long term human health effects from exposure to trace levels of PCDDs and PCDFs are needed. Such studies require methods for accurate assessment of human exposure to these chemicals.

11. Studies attempting to relate PCDD/PCDF formation to incinerator conditions or levels of precursors will probably not be successful unless a truly comprehensive program is established in which hundreds of tests over a lengthy time (say, one year) are taken. Although it is unlikely that the results from such an investigation would justify the expense, some studies are needed in which a single incinerator is studied over a much longer time than has currently been reported. For example, if a link between fly ash concentrations of PCDD/PCDF and stack

emissions could be established, then incinerator emissions could be monitored inexpensively by periodic analysis of fly ash. Such information cannot be obtained by monitoring an incinerator only 2 - 3 times. The accuracy of estimating annual emissions from only a few tests is questionable.

12. Bioanalytical techniques based on in vitro cell cultures to estimate the net toxic effects of PCDDs and PCDFs in extracts from environmental media. Methods of determining 2,3,7,8-T_4CDD toxic equivalents in vitro using cultured cells are needed to add biological realism to existing analytical methods. Such methods may be incorporated into future PCDD/PCDF standards.

13. Development of in situ biological methods to monitor impact points in Ontario are needed. Such methods could compare biological activities and PCDD/PCDF concentrations in locally restrained populations of animals or plants with control populations in unaffected areas.

14. In conjunction with Health and Welfare Canada MOE should undertake a food basket survey to quantify PCDD and PCDF residues in the diet.

15. Further monitoring of levels of PCDDs and PCDFs other than 2,3,7,8-T_4CDD in Ontario fish is required.

2.0 INTRODUCTION

2.1 PHYSICAL AND CHEMICAL PROPERTIES OF PCDDs AND PCDFs

2.1.1 POLYCHLORINATED DIBENZO-P-DIOXINS (PCDDs)

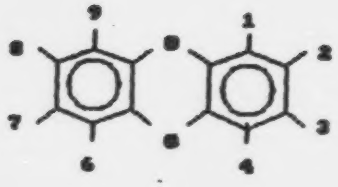
PCDDs are a group of 75 chemically related compounds or congeners identified by the number and position of their chlorine atoms (up to 8 chlorines per molecule). Based on environmental monitoring and analysis of human tissue, it is now known that this family of chemicals are persistent contaminants in terrestrial and aquatic ecosystems. There are 75 different PCDDs whose toxicity ranges from extremely toxic in the case of 2,3,7,8-T_4CDD to some which are many orders of magnitude less toxic.

The scope of PCDD congeners by chlorine substitution group and isomer with the number of isomers shown in brackets is found in Table 2.1.1A.

The physical properties of a few PCDDs are given in Table 2.1.1B. 2,3,7,8-T_4CDD is the best studied PCDD. PCDDs are lipophilic being more soluble in fats and oils than in water. PCDDs are fairly stable chemicals being relatively inert to acids, bases, oxidation, reduction and heat. This chemical stability increases with increasing chlorine substitution. The combination of chemical stability and lipophilicity has resulted in their widespread distribution and persistence in the environment.

Table 2.1.1A

A LIST OF PCDD CONGENERS INDICATING NUMBERS OF ISOMERS IN EACH CHLORINE SUBSTITUTION GROUP. (NUMBER OF ISOMERS)



Numbering system for congeners

M_1-(MONOCHLORO)-(2)

1; 2;

D_2-(DICHLORO)-(10)

1,2; 1,3; 1,4;
1,6; 1,7; 1,8;
1,9; 2,3; 2,7;
2,8;

T_3-(TRICHLORO)-(14)

1,2,3; 1,2,4;
1,2,6; 1,2,7;
1,2,8; 1,2,9;
1,3,6; 1,3,7;
1,3,8; 1,3,9;
1,4,6; 1,4,7;
1,7,8; 2,3,7;

T_4-(TETRACHLORO)-(22)

1,2,3,4; 1,2,3,6;
1,2,3,7; 1,2,3,8;
1,2,3,9; 1,2,4,6;
1,2,4,7; 1,2,4,8;
1,2,4,9; 1,2,6,8;
1,2,6,7; 1,2,7,8;
1,2,6,9; 1,2,8,9;
1,2,7,9; 1,3,6,9;
1,3,6,8; 1,3,7,9;
1,3,7,8; 1,4,7,8;
1,4,6,9; 2,3,7,8;

P_5-(PENTACHLORO)-(14)

1,2,3,4,6; 1,2,3,4,7;
1,2,3,6,7; 1,2,3,6,8;
1,2,3,6,9; 1,2,3,7,8;
1,2,3,7,9; 1,2,3,8,9;
1,2,4,6,7; 1,2,4,6,8;
1,2,4,6,9; 1,2,4,7,8;
1,2,4,7,9; 1,2,4,8,9;

H_6-(HEXACHLORO)-(10)

1,2,3,4,6,7;
1,2,3,4,6,8;
1,2,3,4,6,9;
1,2,3,4,7,8;
1,2,3,6,7,8;
1,2,3,6,7,9;
1,2,3,6,8,9;
1,2,3,7,8,9;
1,2,4,6,7,9;
1,2,4,6,8,9;

H_7-(HEPTACHLORO)-(2)

1,2,3,4,6,7,8;
1,2,3,4,6,7,9;

O_8-(OCTACHLORO)-(1)

1,2,3,4,6,7,8,9;

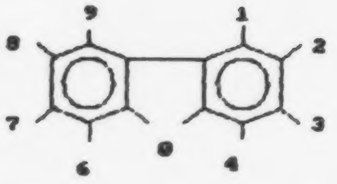
adapted from NRCC (1981a)

TABLE 2.1.1B
PHYSICAL PROPERTIES OF SELECTED PCDDs

	2,3,7,8-T_4CDD	1,2,3,7,8-P_5CDD	1,2,3,6,7,8-H_6CDD	1,2,3,7,8,9-H_6CDD	O_8CDD
Molecular Formula	$C_{12}H_4Cl_4O_2$	$C_{12}H_3Cl_5O_2$	$C_{12}H_2Cl_6O_2$	$C_{12}H_2Cl_6O_2$	$C_{12}Cl_8O_2$
Molecular Weight	322	357	391	391	460
Melting Point (°C)	305	240	285	243	130
Solubilities (g/L)					
o-dichlorobenzene	1.4				1.83
chlorobenzene	0.72				
anisole			2.6		1.73
xylene					3.58
benzene	0.57		1.6		1.0
chloroform	0.37				0.56
n-octanol	0.048				
methanol	0.01				
acetone	0.11				0.005
dioxane					0.38
water	0.0000002				

after Esposito et al, 1980; NRCC, 1981b and U.S. EPA, 1984

Table 2.1.2A
A LIST OF PCDF CONGENERS INDICATING NUMBERS OF ISOMERS IN EACH CHLORINE SUBSTITUTION GROUP. (NUMBER OF ISOMERS)



Numbering system for congeners

M_1-(MONOCHLORO)-(4)

1; 2; 3; 4;

D_2-(DICHLORO)-(16)

1,2; 1,3; 1,4; 1,6;
1,7; 1,8; 1,9; 2,3;
2,4; 2,6; 2,7; 2,8;
3,4; 3,6; 3,7; 4,6;

T_3-(TRICHLORO)-(28)

1,2,3; 1,2,4; 1,2,6; 1,2,7;
1,2,8; 1,2,9; 1,3,4; 1,3,6;
1,3,7; 1,3,8; 1,3,9; 1,4,6;
1,4,7; 1,4,8; 1,4,9; 1,6,7;
1,6,8; 1,7,8; 2,3,4; 2,3,6;
2,3,7; 2,3,8; 2,4,6; 2,4,7;
2,4,8; 2,6,8; 3,4,6; 3,4,7;

T_4-(TETRACHLORO)-(38)

1,2,3,4; 1,2,3,6; 1,2,3,7 1,2,3,8;
1,2,3,9; 1,2,4,6; 1,2,4,7 1,2,4,8;
1,2,4,9; 1,2,6,7; 1,2,6,8 1,2,6,9;
1,2,7,8; 1,2,7,9; 1,2,8,9 1,3,4,6;
1,3,4,7; 1,3,4,8; 1,3,4,9 1,3,6,7;
1,3,6,8; 1,3,6,9; 1,3,7,8 1,3,7,9;
1,4,6,7; 1,4,6,8; 1,4,6,9 1,4,7,8;
1,6,7,8; 2,3,4,6; 2,3,4,7 2,3,4,8;
2,3,6,7; 2,3,6,7; 2,3,7,8 2,4,6,7;
2,4,6,8; 3,4,6,7;

P_5-(PENTACHLORO)-(28)

1,2,3,4,6; 1,2,3,4,7; 1,2,3,4,8; 1,2,3,4,9;
1,2,3,6,7; 1,2,3,6,8; 1,2,3,6,9; 1,2,3,7,8;
1,2,3,7,9; 1,2,3,8,9; 1,2,4,6,7; 1,2,4,6,8;
1,2,4,6,9; 1,2,4,7,8; 1,2,4,7,9; 1,2,4,8,9;
1,2,6,7,8; 1,2,6,7,9; 1,3,4,6,7; 1,3,4,6,8;
1,3,4,6,9; 1,3,4,7,8; 1,3,4,7,9; 1,3,6,7,8;
1,4,6,7,8; 2,3,4,6,7; 2,3,4,6,8; 2,3,4,7,8;

H_6-(HEXACHLORO)-(16)

1,2,3,4,6,7; 1,2,3,4,6,8; 1,2,3,4,6,9;
1,2,3,4,7,8; 1,2,3,4,7,9; 1,2,3,4,8,9;
1,2,3,6,7,8; 1,2,3,6,7,9; 1,2,3,6,8,9;
1,2,3,7,8,9; 1,2,4,6,7,8; 1,2,4,6,7,9;
1,2,4,6,8,9; 1,3,4,6,7,8; 1,3,4,6,7,9;
2,3,4,6,7,8;

H_7-(HEPTACHLORO)-(4)

1,2,3,4,6,7,8; 1,2,3,4,6,7,9;
1,2,3,4,6,8,9; 1,2,3,4,7,8,9;

O_8-(OCTACHLORO)-(1)

1,2,3,4,6,7,8,9;

The use of the term "dioxin" has become synonymous in many peoples minds and in media coverage with the extremely toxic 2,3,7,8-T_4CDD isomer. The term is, in fact, a general one which can refer to the whole of the 75 chemical compounds. In this report the term (PCDD) congener is used when referring to polychlorinated dibenzo-p-dioxins generally. In the case of specific groups of isomers, they are identified as T_4CDD or O_8CDD for example. When a specific isomer, including 2,3,7,8-T_4CDD, is being discussed, it is identified fully as such. Therefore, throughout this report, the most toxic isomer (2,3,7,8-T_4CDD) is specifically identified only in those cases where it has been individually analyzed. It should be noted that most of the literature on animal toxicology has been done in the United States, and deals with 2,3,7,8-T_4CDD because of the special problems with manufacturing and waste disposal from trichlorophenol production facilities particularly in the production there of 2,4,5-T herbicides. As is more fully discussed in Chapter 4, there is no current formulation of 2,4,5-T in Ontario and there has never been any Canadian production of 2,4,5-T precursors which result in highly contaminated still bottom wastes such as those causing the Times Beach, Missouri and Love Canal, Upper New York State contamination problems.

In Ontario there are problems with 2,3,7,8-T_4CDD contamination of fish in Lake Ontario and the Niagara River because of contamination from U.S. sources into these boundary waters. The PCDDs which are being emitted into the environment in Ontario consist of other congeners which are associated with municipal waste and sludge incineration, wood burning and the use of chlorophenol-based pesticides. Chapter 4 discusses these items in more detail.

2.1.2 POLYCHLORINATED DIBENZOFURANS (PCDFs)

PCDFs are a large family of chemical compounds containing 135 different congeners which are chemically similar to PCDDs. Based on analysis of air, water, fish tissue and human tissue in Ontario, it is evident that they are also widely distributed contaminants occurring in some cases at higher levels than PCDDs.

As with PCDDs, the congeners differ in their toxic potential with the toxicity patterns paralleling those of similarily substituted PCDDs but with the acute toxicity being generally 2 to 10 fold lower than the parallel PCDDs. Therefore, some of the isomers, e.g., 2,3,7,8-T_4CDF and 2,3,4,7,8-P_5CDF must be considered extremely toxic. Table 2.2.2A shows the congener, and isomer group distribution of the PCDFs.

2.2 REVIEW OF MAJOR REPORTS ON PCDDs AND PCDFs

As discussed in Chapter 1, the approach taken in the development of this document was to concentrate upon published literature of the subject. In particular, much of this work has built upon earlier reviews prepared by the National Research Council of Canada, Chlorinated Phenols: Criteria for Environmental Quality (NRCC 1982), and the National Research Council of Canada publications on the Polychlorinated Dibenzo-p-dioxins (1981a, b). In addition, in December, 1983, the Departments of

Health and Welfare Canada and Environment Canada jointly released a report to the Federal Ministers of the Federal Ministers' Expert Advisory Committee on Dioxins discussing the concern to Canada of PCDDs and a report of the Interdepartmental Committee on Toxic Chemicals, Dioxins in Canada, the Federal Approach, setting out how the Federal Government wishes to focus its research effort on this problem. Also in preparation at the present time is a report by the National Research Council of Canada on Polychlorinated Dibenzofurans paralleling the excellent review on PCDDs. In the United States, the U.S. Environmental Protection Agency released in December, 1983 a dioxin strategy setting out the priorities for action both for research and clean-up in the United States. This latter document identifies the problem of concern in that country as 2,3,7,8-T_4CDD contamination associated with the past manufacture and disposal of byproducts of 2,4,5-T. As a result, the U.S. focus and priorities are necessarily different than in Canada where this type of activity has not occurred. In fact, the fourth priority out of 7 in the United States for action is PCDDs from incineration sources. By contrast, the information provided in Chapter 4 identifies municipal waste incineration as one of the principal sources of PCDDs in Ontario.

More recently, the U.S. Environmental Protection Agency released scientific criteria documents concerning ambient water quality criteria for 2,3,7,8-T_4CDD (February 1984) and a two-part preliminary draft of a health assessment document for PCDDs (May 1984).

However, as the vast body of research literature on the toxicity of PCDDs is based on 2,3,7,8-T_4CDD toxicity, the authors of this report do depend very heavily on that research. The mixture of PCDDs and PCDFs to which Ontario citizens are exposed in the Ontario environment is vastly different from the 2,3,7,8-T_4CDD contamination which has been experienced in Times Beach, Missouri and in the Love Canal, Upper New York State area. In developing the recommended maximum allowable intake of PCDDs and PCDFs in Chapter 3, estimates were made of the comparative toxicity or toxic equivalents of the various PCDD and PCDF congener groups. This resulted in the introduction of a number of assumptions into the predictions of the risk to humans.

2.3 ABBREVIATIONS

PCDD	Polychlorinated dibenzo-p-dioxin
M_1CDD	Monochlorodibenzo-p-dioxin
D_2CDD	Dichlorodibenzo-p-dioxin
T_3CDD	Trichlorodibenzo-p-dioxin
T_4CDD	Tetrachlorodibenzo-p-dioxin
P_5CDD	Pentachlorodibenzo-p-dioxin
H_6CDD	Hexachlorodibenzo-p-dioxin
H_7CDD	Heptachlorodibenzo-p-dioxin
O_8CDD	Octachlorodibenzo-p-dioxin
DD	Dibenzo-p-dioxin
PCDF	Polychlorinated dibenzofuran
M_1CDF	Monochlorodibenzofuran
D_2CDF	Dichlorodibenzofuran
T_3CDF	Trichlorodibenzofuran
T_4CDF	Tetrachlorodibenzofuran
P_5CDF	Pentachlorodibenzofuran
H_6CDF	Hexachlorodibenzofuran

H_7CDF	Heptachlorodibenzofuran
O_8CDF	Octachlorodibenzofuran
DF	Dibenzofuran
CP	Chlorophenol
T_3CP	Trichlorophenol
T_4CP	Tetrachlorophenol
P_5CP	Pentachlorophenol
NaP_5CP	Sodium pentachlorophenate
2,4-D	2,4-dichlorophenoxy acetic acid
2,4,5-T	2,4,5-trichlorophenoxy acetic acid
tonne	10^3 kilogram
kg	kilogram
g	gram
mg	milligram (10^{-3}g)
ug	microgram (10^{-6}g)
ng	nanogram (10^{-9}g)
pg	picogram (10^{-12}g)
fg	femtogram (10^{-15}g)
dscm	dry standard cubic metre
AHH	aryl hydrocarbon hydroxylase
GLC	gas-liquid chromatography
i.p.	intraperitoneal
i.v.	intravenous
LOEL	lowest observed-effect level
M	male
MFO	mixed-function oxydase
MS	mass spectrometry
NOEL	no observed-effect level
PCB	polychlorinated biphenyl
SCE	sister chromatid exchange
SD	Sprague-Dawley

ADI	acceptable daily intake
bw	body weight
BCF	bioconcentration factor
DNA	deoxyribonucleic acid
ED_{50}	median effective dose
GC/MS	gas chromatography/mass spectrometry
LC_{50}	concentration lethal to 50% of recipients
LD_{50}	dose lethal to 50% of recipients
LOAEL	lowest-observed-adverse-effect level
MFO	mixed function oxidase
NOAEL	no-observed-adverse-effect level
ppm	parts per million
ppb	parts per billion
ppt	parts per trillion
ppq	parts per quadrillion
UV	ultraviolet
WCOT	wall-coated open tabular

2.5 GLOSSARY

accuracy	the extent to which the results of a calculation or the readings of an instrument approach the true values of the calculated or measured quantities and are free from error.
acute toxicity	refers to the effects produced in a short time period by the test material when administered in a single dose.
adenoma	a benign neoplasm of glandular epithelium, also used to describe benign tumours of mucosal epithelium.

alopoecia	loss of hair
bioaccumulation	uptake and retention of environmental substances by an organism from both its environment (i.e., directly from the water) and its food.
bioassay	a determination of the concentration or dose of a given material necessary to affect a test organism under stated conditions.
bioconcentration	the amount of a substance accumulated by an organism by adsorption and by adsorption via oral or other routes of entry, which results in an increased concentration of the substance by the organism or specific tissues.
bioconcentration factor	the ratio of the measured residue in the organism compared to the residue of the substance in the ambient air, water or soil environment of the organism.
biological magnification	the successive increase in levels of a chemical along the food chain.
blepharitis	inflammation of the eyelids.
boultinization	a process of conditioning and treating wood using heat and vacuum.

carcinogen	any agent that initiates development of a carcinoma or any other sort of malignancy.
carcinogenesis	the process of formation of a carcinoma (qv) or any other sort of malignancy.
carcinoma	a malignant epithelial tumour.
chloracne	an acne-like skin eruption caused by halogenated aromatic hydrocarbons.
chronic toxicity	refers to the effects produced by the test material when administered in repeated doses over a long period of time, usually the major portion of the expected life span of short-lived species, sometimes covering the entire life-span and more than one generation of such species.
chromosomal aberration	an abnormal chromosome complement resulting from the loss, duplication or rearrangement of genetic material.
chromosome	a structure in the nucleus of a cell containing a linear thread of DNA, which transmits genetic information to future generations of cells.

clastogenic	relating to the occurrence of chromosomal breaks resulting in gain, loss or rearrangement of pieces of chromosomes.
congeners	compounds belonging to the same chemical group, for example, the dioxin group.
congenital anomalies	abnormalities of the fetus or newborn which are either present at birth or show themselves sometime after birth.
cytochrome	any of the complex protein respiratory pigments occurring within plant and animal cells, that function as electron carriers in biological oxidation.
cytosolic	the fluid cytoplasmic fraction of the cell surrounding the organelles and other insoluble cellular components.
embryo-or fetotoxicity	a deleterious effect on the embryo or fetus.
endocrine gland	an organ whose function is to secrete into the blood or lymph a substance that has a specific effect on another organ.
electrophile	an atom or group of atoms that is electron pair seeking.

eukaryotic	a cell with a definitive nucleus. Also spelled eucaryote.
fibroma	a benign, encapsulated tumour composed principally of fibrous connective tissue.
fly ash	particles of ash emitted within flue gases resulting from the combustion of fuel or other materials.
flue gases	waste gases in the chimney flue or stack that are generated from a combustion process.
gavage	the administration of nutrients or other substances by means of a stomach tube.
gene	the biological unit of heredity, self-replicating and located on a definite position on a particular chromosome.
genotoxic	the property of a substance that reversibly or irreversibly damages genetic material.
hepatic	pertaining to the liver.
hyperkeratosis	hypertrophy (overgrowth) of the cornea. Hypertrophy of the horny layer of the skin.
hyperplasia	increase in cell number causing an increase in the size of the tissue or organ.

isomer	chemical compounds that have identical composition and molecular formulae, but different structures, for example, two different tetrachlorodibenzo-p-dioxins.
lesion	an abnormal change in structure of tissue due to injury or disease.
lymphoma	a tumour of the lymphatic system.
micro-agro-ecosystem	a multispecies system with at least two trophic levels (partly) enclosed for the purpose of agricultural research.
microsome	a fragment of the endoplasmic reticulum; a minute granule of protoplasm.
mitochondria	minute cytoplasmic organelles in the form of spherical granules, short rods or long filaments found in almost all living cells; submicroscopic structure consists of an external membrane system.
mutagenic	inducing a permanent change in genetic material in a cell.
necropsy	to perform an autopsy; a post mortem examination of the body to determine the cause of death.
neoplasm	an aberrant raw growth of abnormal cells or tissues; a tumour.

nosology	the study of disease.
oedema	an excessive accumulation of fluid in the cells, tissue spaces or body cavities due to a disturbance in the fluid exchange mechanism.
oncogenic	substances that cause the formation of tumours.
palpate (v)	to examine by touch gently.
pancytopenia	abnormally low numbers of all the formed elements in the blood.
precision	the ability to make accurate measurements repeatedly.
sarcoma	a malignant tumor arising in connective tissue, and composed of onaplastic cells resembling those of supportive tissues.
soft-tissue sarcoma	a cancer developing in the soft connective tissue of bones, muscles, sinews, etc.
teratogenic	a substance that causes the formation of fetal monstrosities.
translocation	a chromosomal aberration involving an interchange between different non-homologous chromosomes.
tumour	any abnormal mass of cells resulting from excessive cellular multiplication.

vicinal	neighbouring or adjoining.
xenobiotic	relating to external agents, whether physical, biological or chemical, which influence the life processes of cells and organisms.

3.0 TOXICOLOGICAL ASSESSMENT

3.1 INTRODUCTION

The primary aim for establishing environmental standards is to protect the health of Ontario's citizens and their environment from the adverse effects of hazardous contaminants. This process involves estimating the potential for human health hazards and adverse effects on other organisms in the Ontario ecosystem due to the presence of, for example, PCDDs/PCDFs in the environment.

The problem of regulating chemical exposure is complex since, once released, chemicals partition into the major environmental compartments: i.e. air, water and land, according to their physical and chemical properties. An environmental standard should ideally take into account all environmental compartments, all routes of human exposure and all accumulation mechanisms affecting organisms in the Ontario ecosystem.

This document is written as part of the decision - making process whereby the Ministry of the Environment and the public will regulate the presence of PCDDs and PCDFs in the Ontario environment.

In evaluating the environmental health hazard of PCDDs and PCDFs, the objective is to determine the ambient concentration or maximum allowable daily rate of intake which over a lifetime of exposure will result in a non-toxic dose. This approach involves assessment of the human and environmental health risks posed by the presence of these chemicals.

This type of quantitative risk assessment which will be called risk analysis in this document has two main components each of which can be evaluated scientifically; namely, hazard assessment and exposure assessment. The term "hazard" for the purposes of this document refers to the toxic exposure to the chemicals being examined.

Toxicity is an intrinsic property of a chemical related to the dose, i.e., the concentrations and duration and type of exposure that an organism receives, and is not population dependent. The dose may be chronic or acute and the endpoint may range from mild untoward effects to lethality.

The term "exposure" means the dose or level of the chemicals people or organisms come in contact with.

The term "risk" can be mathematically quantified as the probability that an adverse effect will occur in a certain proportion of the population following exposure to a specified level of a chemical. The unit of risk is the lifetime.

In risk analysis, the risk is derived from combining the results of the hazard assessment with the exposure assessment. It should be clear that if either the hazard or the exposure potential of the substance is reduced then the risk is also reduced.

Hazard assessment involves hazard identification and hazard estimation. Hazard identification involves examining a wide array of toxicological data.

Consideration must be given to the most sensitive species or part of the human population. Evidence of irreversible injury to somatic cells, interference with normal physiological functions, or injuries resulting in decreased reproductive success are most pertinent to this hazard assessment process.

Hazard estimation is the subsequent process whereby the magnitude of toxic responses is quantified by selecting critical toxicological criteria and evaluating dose-response data.

In the context of this toxicological assessment, determination of a non-toxic lifetime dose is paramount.

This chapter of the scientific criteria document reviews and summarizes current toxicological knowledge related to PCDDs and PCDFs.

Exposure assessment is detailed in Chapters 4 and 5.

Given a wide range of toxic endpoints and their associated dose or effect levels reported in the literature, there is obviously a need to select those sets of data with sufficient scientific evidence. Risk assessment and subsequent regulatory decisions should ideally be based on well-validated, reliable data. Since toxicity (and risk) is dose-related, the data should indicate the dose range or the range of environmental levels over which adverse responses occur.

Three principal sources of scientific evidence are generally used in hazard assessment:

(1) Human epidemiological data from studies of occupational exposure or exposure of the general population. Lifetime human data studies are considered the strongest evidence to indicate a link between exposure and effects in humans. Such studies rarely determine the magnitude of the dose received. (Section 3.4)

(2) Long-term animal studies using laboratory animals are the major basis for assessing health risk to humans in the absence of reliable human epidemiological data. (Section 3.3.2)

(3) Results from both acute toxicity tests and short-term genotoxicity tests are considered to be the minimal evidence upon which to base a risk assessment.

(a) Acute Toxicity Tests

These tests mainly involve laboratory animals and are characterized by the traditional LD_{50} test. (Section 3.2) Most tests on environmental organisms are in this category. (Section 3.5)

(b) Short-Term Genotoxicity Tests

Traditionally, the *Salmonella* test (Ames 1972) has been used. It has been recommended that at least three classes of assay be required to determine the genotoxicity of a chemical compound. Ashby (1982) suggests the use of a 3-tier system, with the *Salmonella* test as the primary tier and a complementary eukaryotic test in the second tier. Chemicals found negative in the first two tests but having clastogenic properties would be tested in a third-tier cytogenetics in vitro test. Similar protocols have been proposed by the NRC/NAS (1983).

Other supporting evidence, which may be related to the physical and chemical properties, chemical structure and biological activity of the chemical under review, will also be used in the hazard estimation process.

The use of these sources of evidence to estimate risk at different levels of exposure is discussed in Section 3.7.

Most of our knowledge of the toxicology of PCDDs and PCDFs is based on acute and long-term animal studies and short-term genotoxicity studies done on 2,3,7,8-T_4CDD. Toxicological knowledge of the other PCDDs and PCDFs is quite fragmentary. Consequently, this toxicological assessment is based primarily on the scientific database accumulated for 2,3,7,8-T_4CDD. A summary of the toxicology of the other PCDDs and PCDFs is found in Section 3.6.

3.2 MAMMALIAN TOXICOLOGY of 2,3,7,8-T_4CDD

3.2.1 INTRODUCTION

Several comprehensive reviews of the biological, biochemical and toxicological effects of PCDDs and PCDFs exist (NRCC, 1981a; Poland and Knutson, 1982; Kociba and Schwetz, 1982a, b). PCDDs and PCDFs share similar chemical structures, patterns of toxic responses, and possibly a common mechanism of action at the cellular level with other halogenated aromatic hydrocarbons such as the polychlorinated biphenyls (PCBs). This suggests that the structure of individual halogenated aromatics may determine their relative activities and mode of action (Parkinson and Safe, 1981).

The actual mechanisms of toxicity of PCDDs and PCDFs are unclear at the present time. However, following exposure to the more biologically active members of this group of chemicals, a series of common biological and toxicological responses can be observed at the cellular, tissue, organ and organism levels which include (i) a wasting syndrome which is manifested by a progressive weight loss, decreased food consumption, depletion of adipose tissue, decreased levels of thyroxine and O_2 consumption, hypothermia, and progressive weakness in the treated animals; (ii) skin disorders: acneform eruption or chloracne, alopecia, oedema, hyperkeratosis and hypertrophy of the Meibomian glands; (iii) thymic atrophy, lymphoid involution and atrophy and effects on the immune system; (iv) impaired liver function leading to increased liver weight, lipid accumulation and porphyria (resembling porphyria cutanea tarda); (v) altered hematological functions, e.g., levels of triglycerides, cholesterol and hemoglobin in the blood and changes in the numbers of erthrocytes, leucocytes and platelets; (vi) endocrine and reproductive disorders; (vii) modulation of chemical carcinogenesis; (viii) the induction of numerous enzymes including the cytochrome P-448 (or P-450) dependent monooxygenases. 2,3,7,8-T_4CDD causes a wide range of effects some of which are highly species dependent.

Current evidence suggests that a cytosolic receptor protein may be involved in the biological and toxic responses described above. 2,3,7,8-T_4CDD and some related PCDDs and PCDFs are potent inducers of several enzyme systems including diverse hepatic and extrahepatic drug-metabolizing enzymes in some animal species. The rodent hepatic cytochrome P-448 dependent monooxygenases (i.e. aryl hydrocarbon hydroxylase, AHH) is readily induced by 2,3,7,8-T_4CDD. 2,3,7,8-T_4CDD is a unique compound and, induces AHH in genetically inbred C57BL/6 and DBA/2 mice which are responsive and non-responsive to 3-methylcholanthrene, respectively. 2,3,7,8-T_4CDD is 10 to 20 times less potent in the latter strain of mice and the difference in their susceptibility is related to a defect in the regulatory gene of the non-responsive mice which codes for the Ah receptor protein. The 2,3,7,8-T_4CDD cytosolic receptor protein has been identified in several tissues of diverse animal species and Poland and Glover (1980) have proposed that this macromolecule plays a pivotal role in initiating the biologic and toxic effects of PCDDs and related toxic halogenated aryl hydrocarbons.

This section briefly reviews the acute toxicity, pathology and physiological effects, biochemical effects, and immunotoxicity of 2,3,7,8-T_4CDD.

3.2.2 ACUTE TOXICITY

Several reviews of acute lethality data for PCDDs and PCDFs have been published (Poland and Knutson, 1982; Kociba and Schwetz, 1982a, b). Data pertaining to the acute toxicity and related histopathology of 2,3,7,8-T_4CDD in different species are summarized in Tables 3.2.2A and 3.2.2B.

There is wide variation in the dosage required to cause death among mammals. The single dose oral LD_{50} ranges from 0.6ug/kg in male guinea pigs to 5051ug/kg in hamsters. This indicates an approximate 5000-fold range of sensitivity in tested mammalian species. Some variation in the sensitivity of the sex of a particular species has been noted; e.g., male rats and male guinea pigs are more sensitive to 2,3,7,8-T_4CDD than the females of the same species.

The LD_{50} for 2,3,7,8-T_4CDD is based on delayed effects, since the death of the animals does not occur until 20 to 30 days after exposure. This delay in toxic response is unaffected by increasing the magnitude of the dose. Below the LD_{50} dose, the time for signs of toxicity to appear increases.

2,3,7,8-T_4CDD is most toxic following oral administration in those species (e.g., rabbit and hamster) where different modes of administration have been used.

Metabolites of 2,3,7,8-T_4CDD obtained from the bile of 2,3,7,8-T_4CDD-treated dogs appear to be detoxified since the acute toxicity of a mixture of these compounds in the guinea-pig is at least 100-fold lower than that of 2,3,7,8-T_4CDD itself (Poiger, et al., 1982).

TABLE 3.2.2A
ACUTE MAMMALIAN TOXICITY OF 2,3,7,8-T_4CDD

Species, Strain	Sex	Route of Exposure	Single Dose LD_{50} (ug/kg body wt.)	References
Guinea pig, Hartley	M	oral	0.6	Schwetz et al., 1973
	F	oral	2.1	McConnell et al., 1978a
Rat, Sherman	M	oral	22	Schwetz et al., 1973
	M	i.p.*	60	
	F	oral	45	
	F	i.p.	25	
Rat, Sprague Dawley	M	oral	60	Beatty et al., 1978
Weanling	M	oral	60	
	F	oral	25	
Rat, Charles River	Mixed	oral	100	
Rat, Porton	F	oral	90	
Monkey, Rhesus	F	oral	ca.70	McConnell et al., 1978b
Rabbit	Mixed	oral	115	Schwetz et al., 1973
		dermal	272	
		i.p.	252	
Mouse, C57B1/Sch	M	oral	114	Schwetz et al., 1973
Mouse, C57B1/6fh	M	oral	284	McConnell et al., 1978a
Mouse, C57BL/6J		i.p.	132	Olson et al., 1983
Mouse, DBA/2J		i.p.	620	
Mouse, B6D2F/J		i.p.	300	
Dog, Beagle	Mixed	oral	ca.200-300	Schwetz et al., 1973
Hamster, Golden (Syrian)	Mixed	oral	1157	Olson et al., 1980b
	Mixed	i.p.	3000	Henck et al., 1981
	M	oral	5051	Gupta et al., 1973

* i.p. - interperitoneal

TABLE 3.2.2B

ACUTE TOXICITY AND HISTOPATHOLOGY OF 2,3,7,8-T$_4$CDD OR RELATED CHLORINATED AROMATIC HYDROCARBONS: SPECIES DIFFERENCES

SPECIES	ACUTE TOXICITY LD$_{50}$ ug/kg-bw	CHRONIC TOXICITY ug/kg-day	WASTING SYNDROME	HYPERPLASIA AND/OR METAPLASIA							HYPOPLASIA, ATROPHY OR NECROSIS			OTHER	
				(1)	(2)	(3)	(4)	(5)	(6)	(7)	(A)	(B)	(C)	(A1)	(A2)
Monkey	70		x	++	+	++	++		++		+	+	+	+	+
Guinea Pig	0.6- 2.1		x	0		++					+	+	+	0	0
Cow			x	+		++	+			+	+	+	+	0	0
Rat	22- 190	.01-0.1	x	0		0		++	0		+		+	++	0
Mouse	114- 300	0.26-0.4	x	0		0	++		0		+	+	+	+	+
Rabbit	115- 272		x			++			+					++	
Chicken	25- 50		x								+	+	+	+	++
Hamster	1157-5051		x	0	++						+			+	+
Dog	200- 300		x												

x demonstrated effect 0 no effect +,++ demonstrated effect ranked in severity Blank - no data

(1) Gastric mucosa
(2) Intestinal mucosa
(3) Urinary tract
(4) Bile duct and/or Gall Bladder
(5) Lung (focal alveolar)
(6) Skin (chloracne)
(7) Skin (X-disease)

(A) Thymus
(B) Bone Marrow
(C) Testicle

(A1) Liver Lesions
(A2) Oedema

Amalgam from Table 3.2.2A and Poland and Knutson, 1982

3.2.3 PATHOLOGICAL AND PHYSIOLOGICAL EFFECTS

The pathological changes induced by 2,3,7,8-T_4CDD in mammalian systems are outlined in Table 3.2.2B.

Long-term animal studies using products contaminated with 2,3,7,8-T_4CDD are generally confounded by toxic effects of the parent compound (e.g., 2,4,5-T) and are not considered in this review.

The effects observed after administration of 2,3,7,8-T_4CDD vary with dose, length of exposure and species of animal. There is much overlap in the histopathological findings from acute as well as sublethal, chronic treatments.

Animals exposed to acutely toxic levels of 2,3,7,8-T_4CDD exhibit a general wasting syndrome. They show weight loss or reduced weight gain accompanied by a depletion of adipose tissue, and progressive weakness until death. Many of the symptoms observed in acute lethality studies are also observed at higher dose levels in both long-term and lifetime chronic feeding studies. Rats receiving more than 0.05 ug 2,3,7,8-T_4CDD/kg body weight/day exhibited reduced food consumption, decreased body weight gain and increased mortality (Kociba et al., 1976, 1978; Van Miller et al., 1977). Continued ingestion of these high chronic dose levels of 2,3,7,8-T_4CDD caused multiple toxicological effects.

External symptoms associated with toxic exposure to 2,3,7,8-T_4CDD include chloracne and other epidermal changes. Chloracne and dermal hyperkeratosis, oedema and blepharitis are observed in monkeys, rabbits and hairless mice but have not been reported in other strains of mice, rats, hamsters or guinea pigs. Alopecia is observed in dogs and monkeys.

Necropsy findings from both lethal and sublethal animal studies include marked loss of body fat, involution of thymus, spleen, lymph nodes, and lymphoid tissues of the gastrointestinal tract. Various vascular anomalies are observed in rodents and non-human primates. Liver pathology is minor or absent in monkeys, guinea pigs and hamsters. Hyperplasia of the epithelium of the stomach, urinary bladder and intestines, and reduced spermatogenesis have also been noted in rodents and primates exposed to 2,3,7,8-T_4CDD.

Liver changes are the most common finding in rodents, with the effects being more pronounced in females. Hepatocellular changes were observed in rats even at the no observable effect level (NOEL) for survival rate and tumour frequency (0.001 ug 2,3,7,8-T_4CDD/kg body wt./day), at termination of the experiment (104 weeks) (Kociba et al., 1978).

Early findings of thymus atrophy, increased liver weight and hepatocellular damage following 2,3,7,8-T_4CDD exposure were determined at dose levels below those at which body weight effects were apparent.

Rhesus monkeys are very sensitive to 2,3,7,8-T_4CDD: daily oral intake of less than 1ug/kg is lethal to young males (McNulty, 1977). Allen et al. (1977) fed 0.5ug 2,3,7,8- T_4CDD/kg body weight/ day to female Rhesus monkeys for 9 months. Five out of the eight animals died following this diet.

Cellular depletion of bone marrow occurred with increasing exposure to 2,3,7,8-T_4CDD, eventually leading to severe pancytopenia prior to death. Similar effects were observed in other studies of Rhesus monkeys (McConnell et al., 1978b) and guinea pigs (McConnell et al., 1978a).

By comparison, only minor haematological changes were observed in rats exposed to toxic levels of 2,3,7,8-T_4CDD (Kociba et al., 1978).

While exposure to lethal doses (either acute or chronic) results in a syndrome of toxic effects, no single symptom has been clearly identified as reflecting the major cause of death.

3.2.4 BIOCHEMICAL EFFECTS

Exposure to 2,3,7,8-T_4CDD results in a spectrum of biochemical effects (Poland and Knutson, 1982).

3.2.4.1 Enzyme Induction

The increase in levels or activity of enzymes commonly referred to as the microsomal monooxygenase system is a highly sensitive response to chlorinated aromatic hydrocarbon exposure (Poland et al., 1979). This enzyme system, identified in many species, is commonly associated with the liver and liver metabolism, but their function in other tissues is also recognized. Such enzymes are extremely important in both "detoxification" and "toxification" of many xenobiotics (Poland and Knutson, 1982).

Measurement of the induction of microsomal monooxygenase enzymes, in response to exposure of the animal to xenobiotics, has become a valuable tool in the assessment of the toxicity of these compounds. Induction of these enzymes has been shown to be associated with exposure to many chlorinated hydrocarbons; however, one of the most effective inducers is 2,3,7,8-T_4CDD (Poland and Knutson, 1982).

2,3,7,8-T_4CDD alters the activities of several enzyme systems. It is a potent inducer of:

a) the hepatic microsomal cytochrome P.450-dependent monooxygenase system, which metabolizes a large number of xenobiotic compounds including PCDDs and PCDFs;

b) the hepatic mitochondrial enzyme, delta-aminolevulinic acid synthetase, which catalyses the rate-limiting step in haem synthesis and may consequently play a role in the development of porphyria;

c) glutathione-S-transferase, a cytosolic glutathione-conjugating enzyme involved in xenobiotic detoxification;

d) ornithine decarboxylase in some animal species and this enzyme has been directly linked to events in carcinogenesis (Nebert _et_ _al_., 1975);

e) cytosolic DT diaphorase (Beatty and Neal, 1976);

f) microsomal UDP-glucuronosyl transferases (Owens, 1977); and;

g) epidermal transglutaminase activity in mouse skin and this biologic response may be directly related to skin morphological changes (Puhvel et al., 1984).

Enzyme induction is not a toxic response per se; however, it is a very sensitive indicator of PCDD or PCDF exposure. It may be used to estimate activity of other PCDDs and PCDFs with reference to 2,3,7,8-T_4CDD. This aspect of the biological activity of PCDDs and PCDFs is discussed in detail in Section 3.6.

3.2.4.2 Clinical Biochemistry

Numerous clinical biochemical parameters related to blood chemistry and hepatic porphyrin metabolism are altered in animals receiving toxic doses of 2,3,7,8-T_4CDD. These effects are apparently species-specific. Mice exhibit decreased total serum protein level and evidence of porphyria (McConnell et al., 1978a). Rats also exhibit elevated levels of urinary porphyrias at doses as low as 0.01 ug 2,3,7,8-T_4CDD/kg body weight/day. At 0.1 ug 2,3,7,8- T_4CDD/kg body weight/day dose, red blood cell and haemoglobin levels were depressed. White cell differential counts were unaffected, however, and the activity of several serum enzymes was increased (Kociba et al., 1978).

Porphyria or altered serum protein levels were not observed in guinea pigs treated with lethal doses of 2,3,7,8-T_4CDD (McConnell et al., 1978a).

Female rhesus monkeys receiving 0.5ug/kg/day of T_4CDD had unchanged levels of blood urea nitrogen, total serum lipids and serum protein; however, white blood cells, platelets, haemoglobin and haemocount decreased with increasing exposure to 2,3,7,8-T_4CDD. (Allen et al., 1978.)

2,3,7,8-T_4CDD-treated rats have also shown decreased serum thyroxine concentrations (Bastomsky, 1977; Potter et al., 1983). Chemical thyroidectomy protected athyroid rats for 45 days from doses of 2,3,7,8-T_4CDD that were lethal to normal or thyroidectomized-euthroid rats over the same period (Rozman et al., 1984). Sublethal doses of 2,3,7,8-T_4CDD increase the lipid content of liver of rats and mice.

These clinical biochemical effects reflect the toxicity of 2,3,7,8-T_4CDD on a multiplicity of organ systems and can be observed following chronic sublethal exposure. Dose-response information for many of these biochemical effects is lacking.

3.2.5 IMMUNOTOXICITY

As discussed in Section 3.2.4, thymic atrophy is caused by 2,3,7,8-T_4CDD in all mammalian species studied. Depression of other lymphatic tissues (spleen, lymph nodes and bone marrow), especially in guinea pigs and non-human primates, is another common feature of 2,3,7,8-T_4CDD intoxication. Consequently, research has been directed towards understanding the action of 2,3,7,8-T_4CDD on the functioning of the immune system. These studies cover at least four major research areas, namely: susceptibility of animals to bacterial and viral infections, cell-mediated immune function, humoral immune system, and the immune system in humans (NRCC, 1981a).

3.2.5.1 Immunosuppressive Effects of 2,3,7,8-T_4CDD

One of the most potent toxic effects of PCDDs and PCDFs is the general suppression of the immune system. Garattini et al., (1982) studied this phenomenon and found cell-mediated immune responses more resistant to the toxic effects of 2,3,7,8-T_4CDD than the humoral-mediated responses. They found that cellular responses of C57B1/6 mice were not modified by single interperitoneal (i.p.) dose up to 30 ug/kg or repeated oral doses up to 5 ug/kg given weekly for 4 weeks, but that humoral responses were markedly inhibited by single doses as low as 1 ug/kg or repeated doses of 0.5 ug/kg.

They also observed an association between the degree of humoral response and the susceptibility of different mouse strains to AHH enzyme induction by 2,3,7,8-T_4CDD.

Clark et al. (1981) and Clark et al. (1983) have also studied the immunosuppression by 2,3,7,8-T_4CDD and have found the effects at doses much lower than those reported by Garattini et al. (1982). They have also reported that the chemical affects the cell-mediated immune system to a greater degree than the humoral-mediated system. These researchers used i.p. injections of 2,3,7,8-T_4CDD in male C57B1/6 mice, a sensitive strain, and found suppression of cytotoxic T-lymphocytes at doses as low as 4 ng/kg. They were able to determine that the suppression was due to the development of 2,3,7,8-T_4CDD-induced suppressor cells and not due to suppression of T-helper cells, or to the depletion of cytotoxic T-lymphocyte precursor cells. The effects seen at 4 ng/kg were obtained using a combination of exposing the mice to 2,3,7,8-T_4CDD followed by determination of the degree of cytotoxic T-lymphocyte suppression in an in vitro system. The lowest dose at which in vivo effects were observed in the mice was 0.4 ug/kg.

These authors have also reported that challenging, 2,3,7,8-T_4CDD-treated mice with herpes virus II resulted in increased mortality (Clark et al, 1983). However, the data to support this statement are not very clear (Table 3.2.5.1A).

TABLE 3.2.5.1A

EFFECT OF 2,3,7,8-T_4CDD TREATMENT ON MORTALITY FOLLOWING CHALLENGE WITH HERPES VIRUS II
From Clark et al. (1983)

Treatment	Proportion Surviving	Significance[a]
Vehicle control	33/76 (43.4%)	-----
40 ng/kg T_4CDD	15/65 (23.0%)	0.009
0.4 ug/kg T_4CDD	18/64 (28.6%)	0.044
4.0 ug/kg T_4CDD	18/63 (29.6%)	0.051

[a] P value in comparison to control was determined by Fisher's Exact test.

It can be seen from Table 3.2.5.1A that there is no dose response evident and that mortality rates apparently improve with increasing dosage. Also, the statistical analysis used is questionable and the significance column is left uninterpreted in the paper by Clark et al., (1983).

Vecchi et al., (1983) studied the variation in immunosuppressive effects between different strains of mice treated with 2,3,7,8-T_4CDD and observed that humoral antibody production was strongly inhibited in C57B1/6 mice at single doses of 1.2 ug/kg. Less sensitive strains of mice (DBA/2 and AKR) required a dose of 6 ug/kg to bring about suppression of humoral antibody production.

An interesting observation has recently been made by Rizzardini et al., (1983) who have observed that mixtures of 2,3,7,8-T_4CDD and 2,3,7,8-T_4CDF are less toxic to the immune system in mice than 2,3,7,8-T_4CDD alone. These researchers also administered mixtures containing PCDDs and PCDFs extracted from urban incinerator emissions and found them to be less toxic than the pure chemicals.

In summary, work on the immunosuppressive effects of TCDD's is in its early stages of understanding and observations made in specifically bred mice exposed to pure substances cannot yet be extrapolated to man. The study of the effects of exposure to mixtures of TCDD's and TCDF's and of mixtures of similar compounds from incinerator emissions has just begun.

Due to the variability in the responses of 2,3,7,8-T_4CDD-exposed animals and contradictions in the findings of various investigations, it cannot be demonstrated that the immune system is adversely affected at exposure levels lower than those causing other toxic effects.

3.2.6 MATRIX EFFECTS ON UPTAKE (BIO-AVAILABILITY)

Oral and Dermal

PCDDs and PCDFs are often found in the environment adsorbed onto particulate matter such as fly ash, sediment and soil. The degree of adhesion is variable and may be a function of the characteristics of the particulate matter in question. Adhesion may be correlated with the percent carbon in the particle, particle size and particle morphology. Toxicological assessments, in the absence of firm data, tend to assume that particulate-associated PCDDs and PCDFs are 100% biologically available once they enter the body. The studies below examine the effect on uptake of the various PCDD matrices.

Data from the following studies are detailed in Tables 3.2.6B, C, and D which are found in Appendix II. Table 3.2.6A at the end of this section summarizes these studies.

The dermal and intestinal absorption of $[^3H]$ - labelled T_4CDD in the rat, from various sources, was investigated (Poiger and Schlatter, 1980) as shown in Table 3.2.6B (Appendix II). The liver concentration of $[^3H]$-T_4CDD was used as a measure of the uptake of the compound since the liver appears to be the main target organ for the rat. The oral studies show optimal uptake from ethanol suspensions of 2,3,7,8-T_4CDD with decreasing uptake from soil-adsorbed to activated carbon-adsorbed 2,3,7,8-T_4CDD. Also, increased contact time of the 2,3,7,8-T_4CDD with soil further reduces availability.

Similarly, the work of Poiger and Schlatter (1980) (Table 3.2.6B in Appendix II) indicates that combining 2,3,7,8-T_4CDD with soil or activated carbon appears to inhibit the availability, and hence rate of absorption, of 2,3,7,8-T_4CDD.

The induction of acnegenic skin lesions (i.e., inflammation, hyperkeratosis, chloracne) on rabbit ears also showed a similar trend (Table 3.2.6B in Appendix II).

The bio-availability of PCDDs and PCDFs from fly ash was investigated using feeding experiments with rats (van den Berg et al., 1983). Crude fly ash extract, purified fly ash extract, and fly ash were fed to rats at comparable dietary concentrations. As in Poiger's work, considerably lower hepatic levels of PCDDs were observed in rats fed with fly ash compared with those fed with fly ash acetone extract (Table 3.2.6B in Appendix II; van den Berg, et al., 1983).

Furthermore, these hepatic differences were even more pronounced for the more highly chlorinated isomers (i.e., P_5-, H_6- vs T_4CDD), Table 3.2.6C in Appendix II, also from the same work (van den Berg et al. 1983) and quoting hepatic concentrations as a percentage of the adminstered dose, illustrates the following points:

(i) Substantial differences in bio-availability on an isomer specific basis are observable between "fly ash-bound" and "fly-ash-extract" isomers.

(ii) The average retention in the liver of the total dose of each isomer group (e.g., Total T_4CDD) varies from 0.16% for Total T_4CDD to 5.3% for Total H_6CDF.

(iii) Selective uptake of isomers may be caused by isomer- specific absorption as well as isomer-specific metabolism.

In conclusion, these preliminary experiments of van den Berg et al., (1983) corroborate those of Poiger and Schlatter, (1980). Bioavailability is strongly influenced by the matrix. These results are more applicable to the Ontario situation since municipal waste incinerator fly ash is used in the experiments.

In a further study, van den Berg et al., (1984) fed 2.5% HCl-pretreated municipal incinerator fly ash daily to male rats, guinea pigs and Syrian golden hamsters for one, two and three months. Analysis of the livers of these animals for PCDDs and PCDFs indicated the following:

a) only 2,3,7,8-substituted congeners were retained in considerable amounts, in the livers of rats and hamsters.

b) higher levels of PCDFs were retained compared with PCDDs with the corresponding substitution pattern.

c) in the rat and hamster, 2,3,4,7,8-P_5CDF was the compound with highest retention in the liver, recovery of this congener being 14% and 8.5%, respectively. 1,2,3,7,8-P_5CDF showed highest retention behaviour in the guinea pig.

These data suggest that the uptake, retention and metabolism of PCDDs and PCDFs is congener - specific. Furthermore, extended exposure to fly ash in the diet leads to significant uptake of specific congeners.

Preliminary results of a bio-availability study were recently reported by McConnell et al., (1984). Single oral doses of 2,3,7,8-T_4CDD-contaminated soil from Times Beach, MO, were administered to guinea pigs and rats (Table 3.2.6D in Appendix II; McConnell et al., 1984).

Clinico-pathological effects in guinea pigs and hepatic enzyme induction in rats dosed with contaminated soil were similar to effects seen in the comparison group fed with pure 2,3,7,8-T_4CDD in corn oil at similar doses. These studies indicate that soil-bound 2,3,7,8-T_4CDD is biologically available, corroborating other soil-bound 2,3,7,8-TCDD studies (Poiger and Schlatter, 1980). Hepatic microsomal AHH induction was the best indicator of 2,3,7,8-T_4CDD action suggesting that the bioavailability of 2,3,7,8-T_4CDD on soil may be in excess of 50%. However, the soil analysis indicated the presence of PCBs and 2,3,7,8-T_4CDF. The authors acknowledged that "the possibility exists that other contaminants present in the soil might potentiate the action of T_4CDD and could be, in part, responsible for the toxic and enzyme induction effects observed in those studies".

Another recent bio-availability study involved single doses of 2,3,7,8-T_4CDD - contaminated soil from Newark, New Jersey administered to guinea pigs by gavage (Umbreit et al., 1984). Most of the animals in comparison groups treated with 2,3,7,8-T_4CDD in corn oil or 2,3,7,8-T_4CDD added to control soil showed typical symptoms of the wasting syndrome and died between 9 and 31 days after dosing. No animals died from corn oil, control soil or 2,3,7,8-T_4CDD-contaminated soil treatments. Animals were observed for 2 months following dosing and then necropsied. Livers were analyzed for PCDDs at termination of the experiment.

No pathological symptoms of 2,3,7,8-T_4CDD toxicity were observed in animals treated with 2,3,7,8-T_4CDD - contaminated soil at necropsy. The ratio of

2,3,7,8-T_4CDD levels in livers in animals treated with 2,3,7,8-T_4CDD in corn oil to the levels in livers of animals treated with 2,3,7,8-T_4CDD-contaminated soil suggested that bioavailability of 2,3,7,8-T_4CDD from this soil was only about 0.01% (M. Gallo - personal communication).

The bioavailability of PCDDs and PCDFs in soot from a PCB-containing transformer fire at Binghampton State Office Building, New York has been studied. The soot contained 1.2 ug 2,3,7,8-T_4CDD/g, 310 ug total PCDFs (including 48 ug T_4CDFs)/g and 0.5% PCBs. The toxicity of the soot and its benzene extract were compared in feeding studies with guinea pigs, dermal studies with rabbits (Silkworth et al., 1982) and LD_{50} studies using chick embryos (Kaminsky et al., 1984).

The soot matrix, in contrast to the results of Poiger and Schlatter (1980) with activated charcoal only had a relatively minor effect on oral toxicity. Soot or its benzene extract were administered in aqueous methylcellulose. In guinea pigs, the oral LD_{50} values were similar (410 ug soot/kg and 327 soot extract/kg). Chick embryos exposed to soot or the soot extract suspended in corn oil vehicle had an LD_{50} of 0.2 mg soot/egg and 0.1 mg soot extract/egg, respectively. In contrast, rabbits dermally exposed to the soot exhibited no effect, while soot extract produced serious inflammation at the site of application.

A significant vehicle effect was observed when comparing the effect of 2,3,7,8-T_4CDD only in corn oil or aqueous methyl cellulose. In guinea pigs LD_{50}s were 2.5 ug/kg and 19 ug/kg for 2,3,7,8-T_4CDD in corn oil and aqueous methyl cellulose, respectively. In the chick embryo test, the respective LD_{50} values for corn oil and aqueous methyl cellulose were 110-180 ug/egg and 1400 ug/egg.

These experiments show that the soot matrix had a role in reducing dermal bioavailability but not via oral routes of exposure.

Inhalation

No information on bioavailability and subsequent absorption through the respiratory tract was found.

Summary

Based on this review of current bio-availability studies it is not possible to estimate an uptake factor due to the preliminary nature of the studies.

However it is concluded that assuming that 100% of the PCDDs and PCDFs associated with particulate matter, i.e. soil, soot, fly ash and sediments, which are ingested, dermally absorbed or inhaled are all biologically available, represents the "worst case" assumption. It is estimated that this assumption introduces at least a two- to five-fold safety factor in the case of oral exposure, into the exposure assessment. In the case of dermal exposure, this assumption may lead to a several hundred-fold safety factor.

TABLE 3.2.6A
SUMMARY OF ABSORPTION STUDIES

Range of Species	Range of Formulations			Range of Dosing Schedules	Range of Absorptions			References
	Pure TCDD	Soil-Bound TCDD	Fly ash-bound TCDD (and related matrices)		Oral	Dermal	Inhalation	
Rat or Guinea Pig	TCDD in diet or in acetone/ corn oil or in 50% ethanol			Single or repeated dosing up to 42 days	36-86%			Piper, 1973; Fries, 1975; Rose et al., 1976; Nolan et al., 1979; Poiger and Schlatter, 1980.
Rat		Aqueous Suspension of soil/ TCDD		Single dose	16-24%[1,2]			Poiger, 1980; See Table 3.2.6B in Appendix II and Section 3.2.6 on Bioavailability.
Rat or Guinea Pig		Times Beach soil containing 2,3,7,8-T_4CDD		Single dose	50%			McConnell et al., 1984; See Table 3.2.6B in Appendix III and Section 5.2.3 on Bioavailability.
Rat			Aqueous Suspension of activated carbon/ TCDD	Single dose	0-0.07%[1]			Poiger and Schlatter, 1980; See Table 5.2.3A in Appendix III and Section 5.2.3 on Bioavailability.
Rat			Municipal fly ash-bound TCDD in diet	Repeated dosing for 19 days	0.6-1.2%[1]			van den Berg et al., 1983

TABLE 3.2.6A (Continued)

SUMMARY OF ABSORPTION STUDIES

Range of Species	Range of Formulations			Range of Dosing Schedules	Range of Absorptions			References
	Pure TCDD	Soil-Bound TCDD	Fly ash-Bound TCDD (and related matrices)		Oral	Dermal	Inhalation	
Rat	TCDD in methanol					14.8 ± 2.6%[1]		Poiger and Schlatter, 1980
Rat		Aqueous Suspension of soil/ TCDD				0.05-2.2%[1]		Poiger and Schlatter, 1980
Rat			Aqueous Suspension of activated carbon/ TCDD			<0.05%[1]		Poiger and Schlatter, 1980
							No information available	

Percent of administered dose found in hepatic tissue. (It is assumed that hepatic levels are valid estimates of the amount absorbed from both oral and dermal routes.)
Increased contact time of TCDD with soil decreases its absorption.

3.2.7 ABSORPTION, DISTRIBUTION AND EXCRETION (PHARMACOKINETICS)

The major pharmacokinetic processes and their relevance to toxicological assessments are listed in Table 5.2.4A.

TABLE 3.2.7A

Pharmacokinetic Process	Relevance for Toxicological Assessment
Absorption	Indicate extent of absorption of PCDDs and PCDFs Indicate extent of absorption of various matrix (e.g., soil) associated 2,3,7,8-T_4CDD (i.e., bioavailability)
Distribution	Indicate major target organs, storage sites. Indicate whether levels detected through biological monitoring in target organs (e.g., adipose tissue) give reasonable estimates of previous exposure.
Excretion	Estimate the half-life of PCDDs and PCDFs in the body.

Recently, the pharmacokinetics of 2,3,7,8-T_4CDD and 2,3,7,8-T_4CDF were well reviewed (Kimbrough et al., 1984; EPA draft, 1983; Decad et al., 1982) and hence will not be discussed in great detail. Table 3.2.6D in section 5.2.3 summarized oral and dermal absorption studies on experimental animals. Absorption through the respiratory tract has not been studied. Distribution studies of 2,3,7,8-T_4CDD and 2,3,7,8-T_4CDF in various species indicate that the liver and adipose tissue are the major storage sites. In all the following studies involving: guinea pigs (Gasiewicz and Neal, 1979; Decad et al., 1982); rats (Van Miller et al., 1976; Kociba et al., 1978; Decad et al., 1982); hamsters (Olson et al., 1980); and mice (Manara, 1982; Decad et al., 1981), the levels in the liver were higher than in adipose tissue. In monkeys (Van Miller et al., 1976; McNulty et al., 1982; Decad et al., 1982) and humans (Facchetti et al., 1980), this pattern was reversed and adipose tissue levels were higher.

During and after distribution, toxicants can undergo metabolism. The extent and type of metabolism usually governs the rate of excretion. Metabolic studies in various species (Kimbrough et al., 1984; EPA Draft, 1983) have shown that 2,3,7,8- T_4CDD gives rise to monohydroxy, dihydroxy, and monomethoxy metabolites. Administration of 2,3,7,8-T_4CDD metabolites from dogs to rats showed rapid elimination of these compounds by the rats (Weber et al., 1982). Also, administration of 2,3,7,8-T_4CDD metabolites from dogs to guinea pigs showed that the metabolites were at least 100 times less toxic (i.e., acute toxicity, observed from mortality data at 5 weeks) than 2,3,7,8-T_4CDD (Poiger and Schlatter, 1982).

Guinea pigs appear to be unable to metabolize and excrete 2,3,7,8-T_4CDF to any great extent (Decad et al., 1982). Rats metabolize and excrete >99% of the administered 2,3,7,8-T_4CDF dose most rapidly (Decad et al., 1982; Poiger et al., 1984). In mice, 55 to 80% of the excreted 2,3,7,8-T_4CDF was metabolized (Decad et al., 1981). Monkeys metabolize 2,3,7,8-T_4CDF to a lesser extent (Decad et al., 1982).

Most of the following studies (Fries, 1975; Piper et al., 1973; Nolan et al., 1979; Rose et al., 1976; Van Miller et al., 1976; Gasiewicz and Neal, 1979; Olson et al., 1980a) also investigated excretion of 2,3,7,8-T_4CDD from various species.

TABLE 3.2.7B

RATES OF ELIMINAION OF 2,3,7,8-T_4CDD OR 2,3,7,8-T_4CDF IN VARIOUS SPECIES

Species	Dose (ug/kg)	Half-life for Whole Body elimination (days)	Percentage Cummulative Excretion of Dose		Reference
			Feces	Urine	
		2,3,7,8-T_4CDD			
Guinea Pig	2 (i.p.)	30	94	6	Gasiewicz and Neal, 1979
Rat	1 (oral)	31	>99	<1	Rose *et al.*, 1976
Mouse,					
C57BL/6J	10 (i.p.)	17	74	26	Gasiewicz *et al.*, 1981 (unpublished)
DBA/2J	10 (i.p.)	37	70	30	
B6D2F_1/J	10 (i.p.)	17	73	27	
ICR/Ha Swiss	135 (oral)	20	71	3	Koshakji *et al.*,1984
Hamster	650 (i.p.)	11	59	41	Olson *et al.*, 1980a
	650 (oral)	15	—	—	
Monkey	1 (oral)	365	>95	<1	McNulty *et al.*, 1982
		2,3,7,8-T_4CDF			
Guinea Pig	6 (i.p.)	40	4.7	2.3	Decad *et al.*, 1982; Ioannou *et al.*, 1983
Rat	30 (i.v., oral)	<2	61.2	2.0	Decad *et al.*, 1982
Mouse,					
C57BL/6J	30 (i.v.)	2	81.9	12.6	Decad *et al.*, 1981
DBA/2J	30 (i.v.)	4	55.8	19.9	
Monkey	30 (i.v.)	8	71.5	13.2	Decad *et al.*, 1982

Rates of elimination of 2,3,7,8-T_4CDD or 2,3,7,8-T_4CDF in various species are summarized in Table 5.2.4B. The range of half-lives found was 11 - 37 days for single doses of 2,3,7,8-T_4CDD. In the Golden (Syrian) hamster, the least sensitive mammal to the acute toxicity of 2,3,7,8-T_4CDD, excretion occurs readily through both the urine and faeces while in all other species, elimination occurs mainly through the faeces.

For single doses of 2,3,7,8-T_4CDF, elimination half-lives range from < 2 days (rat) to 40 days (guinea pig). Apart from the guinea pig, half-lives for 2,3,7,8-T_4CDF are dramatically shorter than 2,3,7,8-T_4CDD. Again, elimination occurs mainly through the faeces.

Pharmacokinetic properties of major interest are: the effect of single versus chronic doses of PCDDs or PCDFs on persistence; the amount of the administered dose and specific isomers retained; and the target organ or tissue involved.

Animal studies of PCDDs have concentrated exclusively on 2,3,7,8-T_4CDD. In rats, 45% of the total dose was still in the liver, 7 days after a single oral dose of 50 ug 2,3,7,8-T_4CDD/kg (Piper et al., 1973). By comparison the percentage of PCDFs remaining in rat liver, 5 days after single i.p. doses of 1 - 10 mg/ kg was as follows: 2,3,4,7,8-P_5CDF (65 - 96%); 1,2,3,7,8-P_5CDF (44%); 1,2,3,4,7,8-H_6CDF (19.5%); 2,3,4,6,7-P_5CDF (7%) and 2,3,7,8-T_4CDF (3.8%) (Yoshihara et al., 1981).

Longer term studies in which rats were dosed five days a week for seven weeks (Rose et al., 1976) or once a week for 45 weeks (Cantoni et al., 1981) or daily for 2 years (Kociba et al., 1978) indicate that steady state concentrations are reached ranging from 0.7% to 3.0% of the cummulative administered dose. Such experiments have not been performed for PCDFs.

Several estimates of the half-lives of PCDDs and PCDFs in man based on these animal studies discussed above have been proposed, e.g. 100 days for 2,3,7,8-T_4CDD (Gehring - personal communication) and 5 to 12 days for 2,3,7,8-T_4CDF (King et al., 1983). However, evidence from follow-up studies where levels of PCDDs or PCDFs have been monitored in blood or adipose tissues of persons following accidental or occupational exposure (Rappe et al., 1983c; Kunita et al., 1984; Masuda and Yoshimura, 1984) indicate half-lives of PCDDs and PCDFs in humans of about one year. The estimated half-life of 2,3,7,8-T_4CDD in the adipose of monkeys is also about one year (McNulty et al., 1982). As indicated in Section 4.3.4.5 the more toxic 2,3,7,8- substituted congeners are also the slowest to be eliminated.

Summary

Numerous studies have been done on the absorption, distribution, and excretion of 2,3,7,8-T_4CDD in animals and to a much lesser extent other PCDDs. Research on PCDFs focuses primarily on 2,3,7,8-T_4CDF. Some evidence is available on distribution and excretion of PCDDs and PCDFs in humans following accidental or occupational exposure.

Based on studies in animals and man, the major storage sites for PCDDs and PCDFs are in the liver and adipose tissues. In rodents, the levels in the

liver are higher than in adipose tissues; in primates (monkeys and man) the reverse is found.

Metabolism of 2,3,7,8-T_4CDD and 2,3,7,8-T_4CDF varies widely with the animal species studied.

Excretion rates and excretion products have been the subject of a number of studies. The half-life of 2,3,7,8-T_4CDD ranges from 11 to 365 days in various species with excretion of 2,3,7,8-T_4CDD from 1% to 41% in urine and 59% to 99% in faeces. For 2,3,7,8-T_4CDF half-lives range from 2 to 40 days with excretion in urine from 2% to 20% and in faeces from 5% to 82% in different animal species.

In humans, estimates and measurements of half-lives for PCDDs and PCDFs equal or exceed 365 days.

Estimates of absorption rates in humans by oral and dermal exposure routes from different matrices based on this review of the animal pharmacokinetic data are found in Table 3.2.7C.

3.2.8 SUMMARY

In spite of the wide variation in mammalian sensitivity, the toxic effects of 2,3,7,8-T_4CDD are produced at extremely low doses. Toxicity studies in laboratory animals are characterized by progressive wasting, decreased body weight gain, and multiple histopathological effects. The magnitude of the toxic response is dependent on the age, sex and species of experimental animal used. The target organ or tissue affected is also species dependent.

TABLE 3.2.7C

SUMMARY OF PHARMACOKINETIC ESTIMATES

Estimates Deduced from Pharmacokinetic Data	Exposure Route	Area of Potential Applicability in Exposure Scenarios (Section 5.3)
90% of TCDD is absorbed from the GI tract.	Oral	Ingestion of contaminated food, drinking water, surface water, and fish.
10-50% of TCDD adsorbed on soil is absorbed from the GI tract.		Ingestion of soil (e.g., by children) contaminated with PCP or 2,4,5-TCP related industrial waste.
5% of TCDD adsorbed on fly ash is absorbed from the GI tract.		Ingestion of soil contaminated with fly ash deposited from the air near combustion sources or windblown from ashdumps.
20% of TCDD is absorbed through the skin.	Dermal	Dermal exposure to contaminated water and to industrial accident (e.g., PCB fires).
1-10% of TCDD adsorbed on soil is absorbed through the skin.		Dermal exposure to soil contaminated with PCP or 2,4,5-TCP related industrial waste.
1% of TCDD adsorbed on fly ash is absorbed through the skin.		Dermal exposure to soil contaminated with fly ash.
The half-life of TCDD in humans is approx. 1 year. Levels of TCDD in adipose tissue and liver accounts reasonably well for previous exposure in humans and non-human primates.		Using adipose tissue levels, and estimated half-life of T CDD in humans extrapolate back to current levels of exposure (i.e., total daily dose).

3.3 GENETIC TOXICOLOGY, CARCINOGENICITY AND EFFECTS ON REPRODUCTION OF 2,3,7,8-T_4CDD

3.3.1 GENETIC TOXICOLOGY

3.3.1.1 Salmonella/Mammalian Microsomal Mutagenicity Tests

The results of Salmonella mutagenicity testing of 2,3,7,8-T_4CDD have been reported by several authors (Geiger and Neal, 1981; Gilbert et al., 1980; Wassom et al., 1978; Seiler 1973; and Hussain et al., 1972). These results are summarized in Table 3.3.1.1A. Included in this table are the genotypic characteristics of the Salmonella tester strains used in the various studies (Ames et al., 1973).

The results of tests utilizing strains containing the histidine G46 mutation have consistently demonstrated negative results (Table 3.3.1.1A). Strains possessing the G46 mutation respond to mutagenic agents inducing base substitution mutations (i.e., alkylating agents) (Ames et al., 1973). Such results indicate 2,3,7,8-T_4CDD probably does not induce base substitution mutations in bacteria. In addition, strains possessing histidine mutations D3052 (which responds to frame shift mutagens such as activated polynuclear aromatic hydrocarbons) were generally reported to give negative results. Results reported by Seiler (1973) on strain TA1534 and strain TA1531, which showed 2,3,7,8-T_4CDD-induced revertant levels one to two times background, should be considered as insufficient evidence to conclude mutagenicity.

TABLE 3.3.1.1A
SALMONELLA MAMMALIAN MICROSOMAL MUTAGENICITY TEST RESULTS

	STRAIN GENOTYPE													
	G46	TA1975	TA1950	TH1530	TH1535	TA100	TA1531	TH1532	TA1537	TA1978	TA1534	TA1538	TA98	REFERENCE
Histidine mutation	G46	G46	G46	G46	G46	G46	C207	C3076	C3076	C3052	D3052	D3052	D3052	Ames et al., 1973
Lipopoly-saccharide	+	rfa	+	gal	rfa	rfa	gal	gal	rfa	rfa	+	rfa	rfa	Ibid
DNA Repair	+	+	uvrB	uvrB	uvrB	uvrB	uvrB	uvrB	uvrB	+	uvrB	uvrB	uvrB	Ibid
PKM101 (episome)	-	-	-	-	-	+	-	-	-	-	-	-	+	Ibid
2,3,7,8-TCDD														
Spot test + S9					-			-	-			-		McCann, 1978 Wassom et al., 1978
Spot test - S9	-			-			?	+			?			Sailer, 1973
Plate + -S9	-	-	-	-		-		-	-			-	-	Gilbert et al., 1980
Plate + S9					-	-			-			-	-	Geiger + Neal 1981
Plate - S9														Ibid
Suspension assay - S9				-				T	-					Hussain et al., 1972
Fluctuation test + - S9		-	-	-	-	-		-	-	-		-	-	Gilbert et al., 1980
O_8CDD														
Spot test - S9	-			-			-	?			?			Sailer, 1973
Dibenzo-p-dioxin														
Plate Incorp. + - S9					-	-			-			-		Commoner, 1976

+ = positive; ? = questionable response; - = negative; blank = not tested; T = dose toxic to bacterial strain.

A positive mutagenic response induced by 2,3,7,8-T_4CDD in strain TA1532, containing the C3076 histidine mutation, has been reported by Seiler (1973). This strain and the related strain TA1537 contain a histidine C3076 mutation and respond to frame shift mutagenic agents such as 9-amino acridine which cause mutations by intercalating with DNA (Ames *et al*., 1973). While repeated testing of 2,3,7,8- T_4CDD in strains containing the histidine C3076 loci have generally been negative, a second positive result on TA1532 was reported by Hussain *et al*. (1972). Although a report of a positive result by two authors could be considered confirmation of mutagenicity, this interpretation could be in error. Of particular importance is the fact that the positive result reported by Hussain *et al*. (1972) was obtained at a dose toxic to the bacterial strain, which could result in misleading results. In addition, the results of Seiler (1973) were obtained using a spot test which is subject to difficulties of diffusion and potential cell toxicity. Without further clarification of the results, an interference due to bacterial cell toxicity cannot be ruled out. Furthermore, more recent studies failed to confirm this mutagenic response with strain TA1532 or TA1537. This suggests that 2,3,7,8-T_4CDD is non-mutagenic on these strains (Commoner, 1976; McCann, 1978; Wassom *et al*., 1978; Gilbert, *et al*., 1980; Geiger and Neal, 1981: Mortelmans *et al.*, 1984).

When 2,3,7,8-T_4CDD was tested as a potential intercalating agent with calf thymus DNA, it had no effect on the melting temperature, and no spectral changes were observed (unpublished results-cited by Hay (1983)). These results also suggest that 2,3,7,8-T_4CDD is not an intercalating agent.

Testing of dibenzo-p-dioxin (the non-chlorinated congener of 2,3,7,8-T_4CDD) and O_8CDD (the fully-chlorinated congener) has also indicated negative mutagenic results (Seiler, 1973; Commoner, 1976). As seen in the summary Table 3.2.1.1A, O_8CDD did induce an elevated response with tester strains TA1532 and TA1534 in the absence of metabolic activation (S-9) (Seiler, 1973). However, the results with these two strains are considered questionable and further confirmations of these findings have not been reported.

3.3.1.2 Other Bacterial Genotoxicity Tests

Testing of 2,3,7,8-T_4CDD at 2ug/mL in the *Escherichia coli* Sd-4 (streptomycin independence) suspension assay has indicated a positive non-dose-related mutagenic response (Hussain *et al.*, 1972). This response was measured at a dose of 2,3,7,8-T_4CDD which was moderately toxic to the tester strain. Although confirmation of this result has not been reported by other authors, this finding may indicate a positive mutagenic response in the *E. coli* SD-4 assay system. However, information on the specificity and sensitivity of the Sd-4 assay to potentially carcinogenic agents is not at present available.

Hussain *et al.* (1972) also concluded that a weakly positive response was induced by 2,3,7,8-T_4CDD in the *E. coli* K-39 prophage induction assay. This assay, however, indicates DNA damage rather than mutagenicity. On closer examination of these data, the results indicate a significant increase in the phage induction relative to the solvent control (DMSO) but not to the untreated control. In the light of this weak response and the absence of corroborative literature information, the response of 2,3,7,8-T_4CDD in the E. coli K-39 assay could, at best, be considered a "doubtful" positive response.

3.3.1.3 Yeast Mutagenicity and Gene Conversion Assays

Bronzetti et al. (1983) have reported a positive response in yeast (test systems unknown) both in vitro and by a host-mediated assay induced by 2,3,7,8-T_4CDD. This compound at a dose of 10ug/mL resulted in a four-fold increase in revertant and gene-converted cell numbers as well as moderate cell toxicity. This response obtained with metabolic activation should be considered a positive in vitro mutagenic result. A confirmation of this positive result was obtained in a host-mediated assay in mice exposed to 25ug/kg 2,3,7,8-T_4CDD. Although other confirmation of this response in yeast has not been reported, this finding should be considered an indication of mutagenic activity in yeast due to 2,3,7,8-T_4CDD.

3.3.1.4 Summary of Microbial Mutagenicity Tests

The preponderance of information in the literature on microbial mutagenicity testing of 2,3,7,8-T_4CDD suggests that the compound is non-mutagenic. While questionable responses were induced by this compound in Salmonella typhimurium strain TA1532 (Seiler, 1973; Hussain et al., 1972) and in E. coli Sd-4 assay (Hussain et al., 1972), these findings have not been corroborated and, at best, can be considered as "doubtful" mutagenic responses. Similarly, there has been no evidence indicating the formation of DNA adducts by 2,3,7,8-T_4CDD (Section 3.3.1.6). However, the indication of positive mutagenicity in the yeast assay (Bronzetti et al., 1983) cannot be ignored. A final conclusion on the mutagenic potential of 2,3,7,8-T_4CDD in microbial systems cannot be resolved at present. Additional research in systems other than the conventional Ames test is recommended to resolve this question.

3.3.1.5 In Vitro Mammalian Cell Mutagenicity Testing

Tests for the DNA damaging activity of 2,3,7,8-T_4CDD utilizing unscheduled DNA synthesis (U.D.S.) in cultured adult rat hepatocytes (Althaus et al., 1982) and cultured human cells (Loprieno et al., 1982) were negative. While these results indicate that 2,3,7,8-T_4CDD does not cause DNA damage in vitro it is possible that the lack of U.D.S. stimulation was due to inhibition of the DNA repair mechanism. Clarification of this situation requires additional experiments with the appropriate controls.

The ability of 2,3,7,8-T_4CDD to transform tissue culture cells has recently been reported by Hay et al. (1983). The isomers 2,8-D_2CDD and 1,3,7-T_3CDD also demonstrated weak cell transformation ability, while the compounds dibenzo-p-dioxin (DD) and O_8CDD were non-mutagenic. A summary of these results is shown in Table
3.3.1.5A.

Induction of cell transformation by 2,3,7,8-T_4CDD was demonstrated over a dose range of 25 to 250ng/mL (25 to 250ug/kg medium), with a minimum effective dose of 25ng/mL (25ug/kg medium).

The cell transformation assay employed in this study utilized Baby Hamster Kidney fibroblast

(BHK21 C3) tissue culture cells and measured colony formation in soft agar as the phenotypic expression of cell transformation (Styles, 1980). Such growth in soft agar is highly correlated (r^2=0.98) with tumourigenicity of transformed fibroblasts (Styles, 1980; Ts'O, 1979). The cytological processes in this transformation are largely unknown, but it has been postulated that genetic and epigenetic processes are involved (Ts'O, 1979). However, tissue culture strain BHK21 C3 used in the testing of PCDDs is assumed to have undergone epigenetic changes and responded rapidly to induction of transformation by mutagenic agents.

Transformation of fibroblasts, as measured by colony formation in soft agar, is assumed to be induced by a mutagenic event. Phenotypic expression, as measured by growth in soft agar, has been shown to be related in frequency with those measured by biochemical endpoints (i.e., HGPRT-or Na^+/K^+ATPase mutants) [Ts'O, 1979]. In addition, the BHK21 transformation assay was shown in an International Collaborative Study to be 78.6-89.3% accurate in discriminating between chemicals described as carcinogenic and non-carcinogenic based on rodent bioassays (Brookes and de Serres, 1981). In this study, the BHK21 assay was shown to detect certain carcinogens not normally detected in the Salmonella mutagenicity assay.

A more recent study of 2,3,7,8-T_4CDD transforming ability in mouse embryo fibroblast (C3H/10T$^1/_2$) tissue culture cells has been reported (Abernethy et al., 1984). This study reported that T_4CDD did not initiate transformation, hence was not mutagenic to these tissue culture cells. The study demonstrated the ability of T_4CDD, to promote the transformation of these cells by N-methyl-N-nitro-N-nitroso-Guanidine (MNNG). Moreover, the action of T_4CDD on BHK21 cells is not inconsistent with the promotary action of these compounds (Ashby, 1984). Unfortunately, only one other test result using other in vitro mammalian test systems has been reported (Rogers et al., 1982).

TABLE 3.3.1.5A

MUTAGENICITY AND TOXICITY OF PCDDs IN BABY HAMSTER KIDNEY CELLS (BHK 21 C13)[a]

Compound	Mutagenicity Score	Dose Range Tested ug/mL	Minimum Effective Dose ug/mL	Toxicity (LD_{50}) ug/mL
DD	-	0.025-0.25	N.A.	2.5
2,8-D_2CDD	w+	?	?	?
1,3,7-T_3CDD	w+	?	?	?
2,3,7,8-T_4CDD(95%)	+	?	?	?
2,3,7,8-T_4CDD	+	0.025-0.25	0.025	0.025 0.25
O_8CDD	-	0.25-2.5	N.A.	0.25 2.5

a) Adapted from Hay 1983.

DD. . . . . dibenzo-p-dioxin

+	Positive
w+	Weakly Positive
-	Negative
N.A.	Not Applicable
?	Data not given

Such tests should be considered a priority to confirm the mutagenic activity of 2,3,7,8-T_4CDD in mammalian cell cultures and to investigate the relative potency of other isomers in this test system.

The ability of 2,3,7,8-T_4CDD to induce transformations in the BHK21 assay could be considered as an indication of mutagenicity of this compound in mammalian cells in vitro.

Rogers et al. (1982) recently reported that 2,3,7,8-T_4CDD increases methotrexate (MTX^R) and excess thymidine (TdR^R) mutants in L5178Y mouse lymphoma cells. These authors concluded that 2,3,7,8-T_4CDD was a direct-acting mutagen in these cells. However, the commonly used endpoint for mutagenicity in the L5178Y mouse lymphoma cell assay is thymine kinase deficiency (Tk). The two significant endpoints detected in the study of Rogers et al. (1982) are infrequently used and more detailed evaluation of these apparent mutations is required (Clive et al., 1983).

The apparent mutagenicity of 2,3,7,8-T_4CDD in in vitro mammalian cell systems as opposed to the non-mutagenic activity in the Salmonella assay is, at first appearance, conflicting and increases the controversy over the mutagenicity of the compound. However, it must be recognized that a bacterial cell and a mammalian cell are uniquely different in morphology as well as in the complexity of genetic information and thus may respond differently to mutagenic agents. While it has been demonstrated that many carcinogenic agents induce mutagenic changes in both assay types, certain carcinogens, i.e., safrole, hexamethylphosphoramide, and 4-dimethylaminoazobenzene, are detected in the BHK21 assay but not easily by the *Salmonella* assay (Brookes and de Serres, 1981). It should, therefore, be considered that the two *in vitro* assay systems each stand alone as measures of mutagenic activity of a compound. Moreover, the demonstration of apparent mutagenicity of 2,3,7,8-T_4CDD in the BHK21 assay system should be considered indicative of mutagenicity of the compound, particularly when indication of mutagenic activity has also been demonstrated in yeast (Bronzetti *et al.*, 1983) and in the *Escherichia coli* Sd-4 Assay (Hussain *et al.*, 1972).

3.3.1.6 Formation of Nucleic Acid Adducts by 2,3,7,8-T_4CDD

Konderosi *et al.* (1973) concluded that 2,3,7,8-T_4CDD did not form adducts with bacteriophage QB RNA. However, this study apparently tested 2,3,7,8-T_4CDD directly and did not test the formation of adducts by metabolically activated 2,3,7,8- T_4CDD.

The binding of 2,3,7,8-T_4CDD (and theoretical metabolic products) to liver macromolecules *in vitro* in Sprague-Dawley rats was investigated by Poland and Glover (1979). The dose applied was 7.5ug/kg tritiated 2,3,7,8-T_4CDD (corresponding to 0.87mCi radioactive label/kg). The authors concluded that while significant association of 2,3,7,8-T_4CDD with protein was evident, relatively minor amounts associated with RNA and none of this compound associated with DNA. Although the reported investigation of DNA adduct formation by 2,3,7,8-T_4CDD and metabolics is sparse, it appears that the compound does not form adducts with DNA.

TABLE 3.3.1.7A

IN VIVO GENOTOXIC EVENTS ASSOCIATED WITH 2,3,7,8-T_4CDD

SPECIES	ROUTE OF ADMINISTRATION	DOSE ug/kg	DOSES APPLIED (freq)	EXPER. DURATION	GENO-TOXIC EFFECT	END-POINT
1. Rat	Oral	200	1	4 weeks	+?	Multinucleated Cells
2. Rabbit	Cutaneous	?	?	?	+?	Multinucleated cells
3. Rat (male)	Gavage I.P. Gavage	10 5,10,15 20	5(daily 1 1	29 days 4 hours 29 days	- - -	chromosomal aberration
4. Rat (male) Osborne-Mendel (Female)	intubation	0.25,1,2,4 .25,5,2,4	26 bi-weekly	13 weeks	+ +	chromosomal aberrations
5. Human	?	?	?	?	-	chromatid and chromosomal aberration
6. Human 4 adults 13 children	?	?	?	?	-	chromosomal aberrations
7. Human	?	?	acute chronic	?	-	chromosomal aberrations
8. Human 8 adults	? ?	? ?	acute chronic	3 years	-(yr1) +(yr3) +(yr1)	decrease in D-group chromosome satellite
9. Human 15 adults	?	?	?	6-16 mos.	-	chromosomal aberration S.C.E.

TABLE 3.3.1.7A (Con'td)

IN VIVO GENOTOXIC EVENTS ASSOCIATED WITH 2,3,7,8-T_4CDD

TISSUE OR ORGAN	MIN. EFFECTIVE DOSE ug/kg	COMMENT	REFERENCE
1. liver	?	Multinucleated cells may not reflect genetic damage	Greig et al., 1975
2. ?	?	ibid	Kimbrough et al., 1977
3. bone marrow	N.A.		Green & Moreland, 1975
4. bone marrow	2 4	no control group C.A. Pregnancy referenced to those of lowest dose group	Green et al., 1977
5. periph. lymphocytes	?	2,3,7,8-T_4CDD < 0.1mg/kg in 2,3,5-TCPE	Czeizch and Kiraly, 1976
6. ?	N.A.	Test within 2 weeks of exposure -skin lesions -Seveso	Reggiani, 1980
7. ?	?	lesions not reported-Seveso	Mottura et al., 1981
8. lymphocytes	?	significance of increase in satellite associations not known - Seveso	Di Lernia et al., 1982
9. 1 lymphocyte	N.A.	Test-10 yrs. after exposure to "Agent Orange"- skin lesions	Mulcahy, 1980

3.3.1.7 *In Vivo* Mammalian Mutagenicity Tests of T_4CDD

Results of *in vivo* mutagenicity tests on 2,3,7,8-T_4CDD, in which chromosomal damage primarily is the genetic endpoint surveyed, have been reported for a variety of mammalian species including rat, rabbit, and man. A summary of the results of various studies is shown in Table 3.3.1.7A. In general, results of mammalian *in vivo* testing of 2,3,7,8-T_4CDD have produced conflicting results. Positive results have been reported for a variety of systems; however, when positive, the findings have been inconclusive. Generally, scientific evidence to date has been insufficient to conclude mutagenic activity of 2,3,7,8-T_4CDD in *in vivo* mammalian systems.

In vivo testing in rats exposed orally to a 200ug/kg dose of 2,3,7,8-T_4CDD demonstrated the formation of multinucleated hepatic parenchymal cells (Greig *et al.*, 1973). Similar findings in rat systems have been reported by Buo-Hoi *et al.*, (1972) and Gupta *et al.* (1973), and in rabbits following cutaneous exposure to this compound (Kimbrough *et al.*, 1977). Multinucleate cell formation, however, apparently does not reflect genetic damage but rather the association of pre-existing degenerating parenchymal cells (Greig and Osborne, 1978).

Green and Moreland (1977) report no chromosomal aberrations in bone marrow of male rats at 29 days following a single gavage dose of 20ug/kg or after five daily 10ug/kg gavage doses of 2,3,7,8-T_4CDD. Similar negative results were reported in this study in rats sacrificed at 24 hours after single 5, 10, and 25 ug/kg intraperitoneal doses of the compound. In a later study, weak induction of chromosomal aberration in bone marrow of rat following doses of 2ug/kg (male) and 4ug/kg (female) was reported (Green et al., 1977). This later study, however, failed to contain an adequate zero dose control and increased levels of aberrations were related to those of lower-dosed animals. Since the incidence of chromosomal breaks at the optimal dose was 4.65% and considered a weak response, and in the absence of control values, this evidence is insufficient to conclude this compound as clastogenic in rats.

Studies conducted on humans exposed to unknown doses of 2,3,7,8-T_4CDD as a contaminant of various herbicides have also produced inconclusive results. An increased incidence of chromatid and chromosomal aberrations in peripheral lymphocytes of humans exposed to 2,4,5-TCPE and Brominal has been reported by Zceizel and Kraly (1976). However, these findings have not been substantiated in later studies. Mulcahy (1980) failed to detect chromosomal lesions in humans exposed to "Agent Orange". However, this study was performed ten years after the reported exposure, which was associated with skin eruptions, of these individuals. In addition, absence of chromosomal aberration was reported in 17 individuals within two weeks exposure to 2,3,7,8-T_4CDD as a contaminant of trichlorophenol (Reggiani, 1980). An absence of gross chromosomal aberration but a decrease in satellite association with group D-human chromosomes were reported for workers chronically exposed to 2,3,7,8-T_4CDD (DiLernia et al., 1982). Similar decreases in satellite associations to D-group chromosome were reported by these authors at three years but not at one year after acute exposure of eight individuals during the Seveso accident. The relevance of these later findings as possible indicators of genetic damage is presently unknown.

Summary

Results of testing of 2,3,7,8-T_4CDD in mammalian systems have, to date, provided insufficient evidence of chromosomal damaging activity of this compound in *in vivo* systems. While a weak induction of chromosomal aberrations in rats and induction in humans have been reported in isolated studies, the predominance of evidence suggest the compound is not clastogenic. Further studies of this type using other genetic endpoints are necessary to resolve the uncertainty regarding the genetic toxicity of this compound in *in vivo* mammalian systems.

3.3.2 CARCINOGENICITY OF 2,3,7,8-T_4CDD IN MAMMALS

Studies of the carcinogenicity of 2,3,7,8-T_4CDD have been conducted in two mammalian species, rat and mouse. All studies of sufficient duration indicate that chronic administration of low doses of 2,3,7,8-T_4CDD is associated with increased incidence of tumours in these mammals.

Van Miller *et al.* (1977) reported the incidence of neoplasms in male Sprague-Dawley rats. In this study, groups of 10 rats were chronically exposed for 78 weeks to feed containing 1, 5, 50 and 500 ppt as well as 1, 5, 50, 500 and 1,000 ppb 2,3,7,8-T_4CDD. The highest 3 exposure levels (50, 500, and 1,000 ppb) were toxic to all animals after 4 weeks. The occurrence of neoplasia in rats at lower exposures is summarized in Table 3.3.2A. The results demonstrate a variety of tumours induced by chronic doses over the estimated range of approximately 0.001 to 0.3ug 2,3,7,8-T_4CDD/kg/day.

TABLE 3.3.2A

SUMMARY OF NEOPLASTIC ALTERATIONS OBSERVED IN RATS FED SUBACUTE LEVELS OF 2,3,7,8-T_4CDD FOR 78 WEEKS [a]

Level of 2,3,7,8-T_4CDD in feed	Approx.[f] Dose ug/kg/day	No. of Neoplasms[b]	Diagnosis
0	0	0	
1ppt[c]	0.00004	0	
5 ppt	0.0001	6	1 ear duct carcinoma 1 lymphocytic leukemia 1 adenocarcinoma (kidney) 1 malignant histiocytoma (peritoneal)[d] 1 angiosarcoma (skin) 1 Leydig cell adenoma (testes)
50 ppt	0.001	3	1 fibrosarcoma (muscle) 1 squamous cell tumour (skin) 1 astrocytoma (brain)
500 ppt	0.014	4	1 fibroma (striated muscle) 1 carcinoma (skin) 1 adenocarcinoma (kidney) 1 sclerosing seminoma (testes)
1 ppb[e]	0.061		1 cholangiocarcinoma (liver) 1 angiosarcoma (skin) 1 glioblastoma (brain) 2 malignant histiocytomas (peritoneal)[d]
5 ppb	0.3	10	4 squamous cell tumors (lung 4 neoplastic nodules (liver) 2 cholangiocarcinomas (liver
50 ppb	3.0	*	
500 ppb	34	*	
1000 ppb	71	*	

a - Source: Van Miller, Lalich and Allen 1977.
b - 10 animals per group.
c - 1 ppt = 10^{-12}g 2,3,7,8-T_4CDD/g food.
d - Metastases observed.
e - 1 ppb = 10^{-9}g 2,3,7,8-T_4CDD/g food.
f - Estimated dose (ug/kg/day) from approximate weekly doses calculated by NTP Technical Report-NTP-80-32 (NIH Publication No. 32-1757).
* - animals died within 4 weeks.

TABLE 3.3.2B

SUMMARY OF SIGNIFICANT TUMOUR INCIDENCE IN MALE SPRAGUE-DAWLEY RATS MAINTAINED ON DIETS CONTAINING 2,3,7,8-T_4CDD[a]

Time Interval of Study	Month 0-12				Month 13-24				Terminal Kill				Total			
Dose level (ug/kg/day)	Control	0 . 001	0.01	0.1	Control	0.001	0.01	0.1	Control	0.001	0.01	0.1	Control	0.001	0.01	0.1
Number of rats examined	5	1	0	4	65	38	46	41	15	11	4	5	85	50	50	50
Rats with Tumours/tumour-like lesions																
Tumours significantly (P<0.005) greater than controls																
Carcinoma (stratified squamous cell) hard palate or nasal turbinates	0	0	0	0	0	0	0	4	0	0	0	0	0	0	0	4[b]
tongue	0	0	0	0	0	1	1	3	0	0	0	0	0	1	1	3[b]
Adenoma of adrenal cortex	0	0	0	0	0	0	2	5	0	0	0	0	0	0	2	5[b]
Tumours significantly less than controls																
Fibroadenoma/fibroma/ lipoma (subcutaneous)	1	0	0	0	8	1	4	6	1	0	1	0	10	1[c]	5	6
Acinar adenoma of pancreas	0	0	0	0	11	3	4	2	3	4	1	0	14	7	5	2[c]
Pheochromocytoma of adrenal	0	0	0	0	19	3	10	3	9	3	0	1	28	6	10	4[c]
Total tumours not significantly different from controls	2	0	0	1	56	17	23	29	28	8	2	6	86	25	25	36
Total tumours	3	0	0	1	94	25	44	52	41	15	4	7	138	40	48	60

[a] - adapted from Kociba *et al.*, 1978;
[b] - significantly greater than control ($p < 0.05$);
[c] - significantly lower than control ($p < 0.05$)

The study of Van Miller et al. (1977) has several shortcomings. In particular, the number of animals per group was small, only males were used and the duration of the bioassay was shorter than usual.

The findings of zero tumour incidence in the control group (0ppt dose) is not normal for this strain of rat. Furthermore, an expected dose-related increase in tumour incidence was not shown. In addition, the diversity of types of neoplasms reported in this study have not been confirmed in later carcinogenicity studies of this compound. Nevertheless, this study indicates an association between 2,3,7,8-T_4CDD and induced neoplasms in Sprague-Dawley rats.

More detailed carcinogenicity studies in rats have been reported by Kociba et al. (1978) and the National Toxicology Program (NIH Publication No. 82-1765, 1982). Both studies show an increase in certain neoplasms associated with chronic exposure to 2,3,7,8-T_4CDD.

Kociba et al. (1978) reported a study of the chronic toxicity of 2,3,7,8-T_4CDD to male and female Sprague-Dawley rats. Groups of 100 rats (50 males and 50 females) were maintained for up to 2 years (104 weeks) on diets supplying 0.1, 0.01 and 0.001ug 2,3,7,8-T_4CDD/kg/day. 2,3,7,8-T_4CDD (99% pure) was dissolved in reagent-grade acetone and mixed thoroughly with control feed to give desired dose levels. In addition, a control group of 172 rats (86 males and 86 females) were maintained for a similar time period on a diet containing control feed and a proportionate quantity of the vehicle, acetone.

TABLE 3.3.2C

SUMMARY OF SIGNIFICANT TUMOUR INCIDENCE IN FEMALE SPRAGUE-DAWLEY RATS MAINTAINED ON DIETS CONTAINING 2,3,7,8-T_4CDD^a

Time Interval of Study	Month 0-12				Month 13-24				Terminal Kill				Total			
Dose level (ug/kg/day)	Control	0.001	0.01	0.1	Control	0.001	0.01	0.1	Control	0.001	0.01	0.1	Control	0.001	0.01	0.1
Number of rats examined	1	2	2	9	60	32	34	36	25	16	14	4	86	50	50	49
Rats with tumours/tumour-like lesions																
Significantly (P < 0.05) greater than controls																
Hyperplastic nodules -Hepatocellular	0	0	0	0	2	1	8	20	6	2	10	3	8	3	18^b	23^b
Carcinoma-Hepatacellular	0	0	0	0	0	0	0	10	1	0	1	1	1	0	1	11^b
Carcinoma (stratified squamous cell) of hard palate or nasal turbinates	0	0	0	0	0	0	1	4	0	0	0	0	0	0	1	4^b
Carcinoma (keratinizing squamous cell) of lung	0	0	0	0	0	0	0	7	0	0	0	0	0	0	0	7^b
Significantly less than control																
Benign tumour - uterus	0	0	0	2	16	5	7	3	12	7	4	2	28	12	11	7^c
Benign neoplasm - mammary gland	0	0	1	0	50	24	23	22	23	11	12	2	73	35	36	24^c
Carcinoma - mammary gland	0	0	0	0	5	3	1	0	6	1	5	0	11	4	6	0^c
Adenoma - pituitary	0	0	0	0	26	12	8	12	17	6	5	0	43	18	13	12^c
Total not significantly different from controls	0	0	0	0	45	12	9	31	24	12	8	0	69	24	17	31
Total tumours/tumour-like lesions	0	0	1	2	144	37	58	109	89	39	45	8	233	76	104	119

[a] - adapted from Kociba _et al._, 1978;
[b] - significantly greater than controls ($p < 0.05$);
[c] - significantly lower than controls ($p < 0.05$)

All rats were palpated monthly and numbers of rats bearing palpable masses recorded. All rats dying or culled during the study were subjected to gross pathological examinations. Necropsies were conducted at the end of the 105th week of the study. All rats were subjected to an histological examination of an extensive list of tissues. Such tissues included those of possible target organs, as well as gross lesions suggestive of tumour formation.

The results of this study are summarized in Tables 3.3.2B and 3.3.2C. These tables demonstrate significant changes in certain tumour types induced by 2,3,7,8-T_4CDD. In male Sprague-Dawley rats at the highest dose a significant increase in stratified squamous cell carcinomas of the hard palate, nasal turbinates or tongue were reported, as well as adenomas of the adrenal cortex. Such tumours appeared during the 13th month of the 24-month study. (Table 3.3.2B.) Other tumours, notably subcutaneous fibroadenoma, fibroma or lipoma, as well as pancreatic acinar adenomas and adrenal pheochromocytoma were significantly lower in the 2,3,7,8-T_4CDD dosed groups than in the control groups. Numerous other tumour types were reported, but none of these were significantly different in numbers from control to dosed groups.

Female Sprague-Dawley rats exposed to T_4CDD doses of 0.01 and 0.1 ug/kg/day demonstrated a significant increase in hepatocellular hyperplastic nodules, which are considered to be pretumourous lesions. (Table 3.3.2C.) In addition, hepatocellular carcinomas, keratinizing squamous cell carcinomas of the lung and stratified squamous cell carcinomas of the hard palate and nasal turbinates were significantly higher, relative to the control, in the highest dose group. The latter of these carcinomas were similar to those reported for male rats. Again, as with male rats, tumour appearance occurred in the later stage of the study (13 to 24 months).

In female rats, other tumours (specifically, benign tumours of the uterus and mammary gland, as well as carcinomas of mammary gland and adenoma of pituitary gland) were significantly lower at the highest dose than the control group. Other tumours were similar in numbers in both control and dosed groups.

In both male and female animals, chronic exposures to 2,3,7,8-T_4CDD did not significantly increase the incidence of total tumours but altered the incidence of specific tumour types. In male rats, 138 tumours in 85 animals in the control group were reported compared to 40, 48 and 60 tumours per 50 animals at doses of 0.001, 0.01 and 0.1ug T_4CDD/kg/day respectively. Total tumour incidence at the highest dose was less than controls (Table 3.3.2B). Similarly, with female rats, tumour incidences at all doses did not exceed that of the control (Table 3.3.2C). In addition, as previously noted, the vast majority of tumours, in both sexes, in control and dosed groups appeared in the latter half of the study.

The National Toxicology Program (1982a) reported on the carcinogenicity testing of 2,3,7,8-T_4CDD in Osborne-Mendel rats. Groups of 50 male and 50 female rats were each exposed to a theoretical dose of 0.1, 0.05 or 0.5ug/kg/wk. (0.0014, 0.007 or 0.07ug/kg/day) 2,3,7,8-T_4CDD. The test compound, dissolved in 9:12 corn-oil:acetone, was administered by gavage. Dosed animals were

TABLE 3.3.2D
SUMMARY OF SIGNIFICANT TUMOUR NUMBERS IN MALE OSBORNE-MENDEL RATS ADMINISTERED 2,3,7,8-T_4CDD BY GAVAGE[1]

	Untreated Control	Vehicle Control	Low Dose	Mid Dose	High Dose	Dose Related Response
Design Dose (ug/kg/day)	0.000	0.000	0.0014	0.007	0.07	
Number of Rats Examined	75	75	50	50	50	
Carcinoma or Adenoma-Thyroid Follicular Cell	8	1 (104)[2]	5^c (56)	8^c (99)	11^a (77)	S.D.
Fibroma - Subcutaneous	3	3 (75)	1 (107)	3 (93)	7^c (70)	S.D.
Adenoma - Adrenal-cortical cell	8	6 (87)	9 (72)	12^b (70)	9 (84)	N.S.D.
Total Primary Tumours	73	57	38	53	55	
Total Animals with Tumours	48	40	29	33	32	
Incidence of Primary Tumours	0.97	0.76	0.76	1.06	1.10	

[1] From NIH No. 82-1765 (NTP, 1982a)

[2] - time tumour first appeared (weeks)

S.D. - significant dose response.

N.S.D. - no significant dose response.

[a] - significantly greater than control ($p < 0.001$)

[b] - significantly greater than control ($p < 0.02$)

[c] - significantly greater than control ($p < 0.05$)

TABLE 3.3.2E

SUMMARY OF SIGNIFICANT TUMOUR NUMBERS IN FEMALES OSBORNE-MENDEL RATS ADMINISTERED WITH 2,3,7,8-T_4CDD BY GAVAGE[1]

	Untreated Control	Vehicle Control	Low Dose	Mid Dose	High Dose	Dose Related Response
Design Dose (ug/kg/day)	0.000	0.000	0.0014	0.007	0.07	
Number of Rats Examined	75	75	50	50	49	
Adenoma or Thyroid Follicular Cell	0	3 (90)	2 (105)	1 (104)	6 (95)	S.D.
Neoplastic Nodules - Liver	1	5 $(90)^{2}$	1 (104)	3 (106)	12^{b} (85)	S.D.
Neoplastic Nodules or Carcinoma - Liver	2	5 (90)	1 (104)	3 (106)	14^{a} (61)	S.D.
Fibrosarcoma - Subcutaneous	-	0 (-)	2 (55)	3 (58)	4^{c} (95)	N.S.D.
Adenoma NOS - Pituitary	0	1 (104)	5^{c} (76)	2 (104)	3 (99)	N.S.
Adenoma or Carcinoma Adrenal	10	11 (77)	9 (65)	5 (83)	14^{c} (85)	S.D.
Total Primary Tumours	76	82	62	51	90	
Total Animals with Tumours	50	54	40	36	43	
Incidence of Primary Tumours	1.01	1.09	1.24	1.02	1.84	

[1] From NIH No. 82-1765 (NTP, 1982a)

2 - time tumour first appeared
SD - significant dose response
N.S.D. - no significant dose response
[a] - significantly different to control ($p < .001$)
[b] - significantly different to control ($p < .017$)
[c] - significantly different to control ($p < .05$)

compared with groups of 75 animals of each sex administered with the solution (vehicle control) as well as groups of 75 untreated animals of each sex. Animals were studied for a minimum of 104 weeks.

Animals were observed regularly for clinical signs and mortality. Necropsied animals were subjected to gross and microscopic examination of major tissues, organs and gross lesions. The incidences of neoplasia (expressed as lesions at a specific anatomical site relative to numbers of animals in which the site was examined) for each dose group were statistically compared to those of the vehicle control group.

The results of the study are summarized in Tables 3.3.2D and 3.3.2E. In male rats, levels of carcinoma or adenoma of thyroid follicular cells, subcutaneous fibroma and adrenal cortical cell adenoma were significantly higher than those of the vehicle control (Table 3.3.2D). The authors concluded, however, that, based on a statistical analysis of dose response and acceptance of the Bonferroni inequality criteria, only the increased incidence of thyroid tumours in these male rats could be related to administration of 2,3,7,8-T_4CDD. In male rats, the incidence of these tumours at doses of 0.007 and 0.07 ug/kg/day were considered significantly different from controls. The 0.0014 ug/kg/day dose was not significantly different from controls.

In female rats, incidence of adenoma of the thyroid follicular cells, liver neoplastic nodules, liver carcinoma, subcutaneous fibrosarcoma, adenoma of pituitary and adenoma or carcinoma of the adrenals were significantly higher than control in at least one of the 2,3,7,8-T_4CDD dosed groups (Table 3.3.2E). The author concluded, however, that only liver lesions were statistically related to 2,3,7,8-T_4CDD.

In the NTP study as well as in the study of Kociba et al. (1978), the appearance of tumours in animals dosed with 2,3,7,8-T_4CDD occurred at a later stage of the animal's life span. Although certain tumours appeared somewhat earlier in 2,3,7,8-T_4CDD dosed groups, generally tumours in dosed and control groups appeared at similar times and not sooner than the second year of the study.

Results of both the NTP (1982a) and Kociba et al. (1978) studies demonstrate similar incidences of total tumours in dosed relative to control groups in male rats. Thus 2,3,7,8-T_4CDD does not appear to induce an increase in total tumours in male rats. However, the NTP study differs from that of Kociba et al. when tumours in female animals are compared, the former demonstrating an increased incidence of total tumours with increasing dose of 2,3,7,8-T_4CDD which was not shown in the Kociba et al. study.

Carcinogenicity bioassays of 2,3,7,8-T_4CDD in mice by dermal and gavage administration have been reported by the National Toxicology Program (NIH Publication No. 82-1765 and No. 82-1757, 1982a, b). Both studies show an association between 2,3,7,8-T_4CDD administration and increased incidence of certain tumours in mice.

Carcinogenicity bioassay of 2,3,7,8-T_4CDD in mice administered 2,3,7,8-T_4CDD by gavage was reported in NIH Publication No. 82-1765 (National Toxicology Program, 1982a). In this study, groups of 50 male B6C3F1 mice were administered doses of 0.01, 0.05 or 0.5ug 2,3,7,8-T_4CDD/kg/week (0.0014, 0.007 or 0.07ug/kg/day). Groups of 50 female mice were given 0.04, 0.2 or 2.0ug/kg/week (0.006, 0.03 or 0.3ug/kg/day) 2,3,7,8-T_4CDD suspended in 9:1 corn oil-acetone was administered by gavage. Groups of 75 male or female received vehicle while groups of 75 animals of each sex were included as untreated controls.

The results of this carcinogenicity bioassay in mice are summarized in Tables 3.3.2F and 3.3.2G. These results generally indicate 2,3,7,8-T_4CDD was carcinogenic in male and female B6C3F1 mice.

In male mice, numbers of hepatocellular adenoma or carcinoma were significantly higher in the high dose group than the vehicle control group. In addition, numbers of these tumours increased with increasing dose of 2,3,7,8-T_4CDD (Table 3.3.2F). The authors concluded an association between 2,3,7,8-T_4CDD administration and these tumours. Moreover, the appearance of first tumours (week 34) at the high dose group was considerably sooner than that in the control group (week 85).

In male mice a dose-related trend in numbers of alveolar/ bronchiolar adenoma or carcinoma was also seen. However, because the incidence of these tumours in dosed groups was not significantly higher than those in the control group, the authors did not consider these associated with 2,3,7,8-T_4CDD administration.

TABLE 3.3.2F

SUMMARY OF NUMBER OF SIGNIFICANT TUMOURS IN MALE B5C3F1 MICE ADMINISTERED 2,3,7,8-T$_4$CDD BY GAVAGE[1]

	Vehicle Control	Low Dose	Mid Dose	High Dose	Dose Related Response
Design Dose (ug/kg/day)	0.000	0.0014	0.007	0.07	
Number of animals per group	75	50	50	50	
Hepatocellular Adenoma or Carcinoma	15 (85)#	12 (71)	13 (79)	27* (34)	S.D.
Alveolar/Bronchiolar Adenoma or Carcinoma	10 (86)	2 (84)	4 (58)	13 (86)	S.D.
Total Primary Tumours	55	32	32	58	
Total Animals with Tumours	40	25	27	37	
INCIDENCE OF TOTAL TUMOURS	0.73	0.64	0.64	1.16	

[1] From NIH Publication No. 82-1765 (NTP, 1982a)

* Significantly different from control (P<0.001)

SD- Significant dose response (P<0.004)

# Weeks to appearance of first tumour of that type.

TABLE 3.3.2G

SUMMARY OF NUMBERS OF SIGNIFICANT TUMOURS IN FEMALE B6C3F1 MICE ADMINISTERED 2,3,7,8-T_4CDD BY GAVAGE[1]

	Vehicle Control	Low Dose	Mid Dose	High Dose	Dose Related Response
Design Dose (ug/kg/day)	0.000	0.0067	0.03	0.30	
Number of animals per group	75	50	50	50	
Hepatocellular Adenoma or Carcinoma	3 (91)#	6 (105)	6 (105)	11* (99)	S.D.
Follicular-Cell Adenoma (Thyroid)	0	3 (90)	1 (107)	5* (99)	S.D.
Histiocytic Lymphoma	18 (70)	11 (89)	13 (85)	20* 99)	S.D.
Lymphoma or Leukemia	18 (76)	12 (89)	13 (85)	20* (40)	S.D.
Subcutaneous Fibrosarcoma	1 (95)	1 (105)	1 (98)	5 (88)	S.D.
Total Primary Tumours	42	32	34	52	
Total Animals with Tumours	36	25	28	34	
INCIDENCE OF TOTAL TUMOURS	0.56	0.64	0.71	1.10	

1 - From NIH Publication No. 82-1765 (NTP, 1982a)
SD - Significant dose response ($P<0.017$)
* - Significantly different from control ($P<0.017$)
# - Weeks to appearance of first tumour of that type

In female mice, numbers of hepatocellular adenoma or carcinoma, follicular-cell adenoma of thyroid, and histocytic lymphoma increased with dose of 2,3,7,8-T_4CDD administered, and all these tumours at the highest dose group were significantly higher than controls (Table 3.3.2G). Numbers of lymphoma or leukemia and subcutaneous fibroma also increased with dose of 2,3,7,8-T_4CDD but were not significantly higher than those in control groups. The authors concluded and that, on the basis of statistical considerations, lymphomas and leukemias might be associated with 2,3,7,8-T_4CDD administration.

In general, liver and thyroid tumours in these animals appeared at times similar to controls. The notable exceptions were lymphomas and leukemias, which in the high dose group appeared at 40 weeks and much earlier than those in the control group.

2,3,7,8-T_4CDD administration in female B6C3F1 mice demonstrated a pronounced dose-related effect on total tumour incidence in these animals (Table 3.3.2G). These dose effects were not seen in male mice, with only the highest dose demonstrating an elevation in total tumour incidence (Table 3.3.2G).

A carcinogenicity bioassay of 2,3,7,8-T_4CDD by dermal administration was reported in NIH Publication No. 82-1757 (National Toxicology Program, 1982b). Acetone suspensions of 2,3,7,8-T_4CDD were applied to clipped backs of 30 male and female Swiss-Webster mice at frequencies of 3 days per week for 99 or 104 weeks. Female mice received 0.005ug 2,3,7,8-T_4CDD per application (approximately 0.11ug/kg/day) while males received 0.001 ug per application (approximately 0.02ug/kg/day). Groups of 45 animals of each sex received vehicle (acetone) by dermal application, while groups of 30 animals of each sex served as untreated controls.

The authors of the study concluded that 2,3,7,8-T_4CDD at 0.02ug/kg/day dermal application was not carcinogenic in male mice, 2,3,7,8-T_4CDD administration at 0.11ug/kg/day to female mice induced a significant increase in integumentary fibrosarcoma. These results indicated 2,3,7,8-T_4CDD was carcinogenic in female Swiss-Webster mice.

In females, 2,3,7,8-T_4CDD administration shortened the time to first appearance of fibrosarcoma from 96 weeks to 56 weeks. A similar reduction in time of this tumour on 2,3,7,8-T_4CDD dosing was also seen in males although the incidence of this tumour in control relative to dosed groups was not significantly different. However, a latency period of greater than one year was necessary for these 2,3,7,8-T_4CDD-induced tumours in female animals.

Summary

Composite data from several studies have demonstrated an association between 2,3,7,8-T_4CDD administration and increased incidences of specific tumour types in mammalian carcinogenicity bioassays. The compound has been demonstrated carcinogenic in Sprague-Dawley and Osborne-Mendel rats as well as in B6C3F1 mice when administered by oral or gavage routes. This compound was also shown to be carcinogenic in female Swiss-Webster mice dosed by a dermal route.

In general, 2,3,7,8-T_4CDD administration was associated with relatively few histologically different tumour types. When administered by the oral route, increased incidence of liver neoplasia was seen in female rats and mice and in male B6C3F1 mice. Other tumours of adrenal cortex, thyroid follicular cells, hard palate and nasal tumours, lung and possibly lymphoma/leukemia are reported but with no consistency between sex or animal species. When administered by the dermal route, the only study suggests an association between 2,3,7,8-T_4CDD and increased incidence of subcutaneous fibrosarcoma.

In all bioassays examined, the onset of tumours associated with 2,3,7,8-T_4CDD administration appeared late in the study and generally at similar times to control groups. The noted exceptions were liver tumours in male B6C3F1 mice and lymphomas or leukemias in female mice of this strain which appeared much earlier than those in controls.

In general, all carcinogenicity bioassays in both rats and mice are associated with relatively high incidence of spontaneous tumours. All bioassays were able, with appropriate statistical analysis, to associate specific tumour induction with 2,3,7,8-T_4CDD administration. However, when reviewed on the basis of total tumour incidence, 2,3,7,8-T_4CDD-dosed groups of both mammalian species demonstrate similar tumour indices as the control. The noted exceptions are those results with the highest dosed group in female Osborne-Mendel rats and male and female B6C3F1 mice, which demonstrated induced total tumour indices appreciably higher than controls.

In general, relatively low doses of 2,3,7,8-T_4CDD are sufficient to induce tumours in mouse and rat bioassays. Carcinogenicity bioassays by the oral route in these two species were conducted over the range of 0.001 to 0.3ug 2,3,7,8-T_4CDD/kg/day. Significant tumour induction in both species was generally demonstrated over the range of 0.007 to 0.3ug 2,3,7,8-T_4CDD/kg/day. In dermal bioassays in mice, a chronic dose of approximately 0.1ug/kg/day was sufficient to induce tumours in these animals.

The results of the carcinogenicity bioassays are not useful in resolving whether this compound acts as an initiator (a compound which triggers tumour initiation through direct action on genetic mechanisms) or whether it functions as a promoter (a compound which assists previously initiated cells to develop into a tumour). Carcinogenicity bioassays designed to resolve this question have drawn conflicting conclusions as to this compound's action as an initiator or promoter. (DiGiovanni et al., 1977, DiGiovanni, et al., 1979; Berry et al., 1978; Kouri et al., 1978; Cohen et al., 1979; Poland et al., 1982 and Pitot et al., 1980).

2,3,7,8-T_4CDD has been tested several times using the two-stage system of mouse skin carcinogenesis. DiGiovanni et al., (1977) obtained no significant data using 2,3,7,8-T_4CDD on female CD-1 mice initiated with 7,12-Dimethyl-benzanthracene (DMBA). Subsequent studies showed that 2,3,7,8-T_4CDD decreased the incidence of papillomas in female CD-1 mice initiated with DMBA or benzo(a)pyrene (BP) (DiGiovanni et al., 1979). Cohen et al., (1979) showed that 2,3,7,8-T_4CDD pretreatment of female Sencar mice suppressed the incidence of DMBA- or BP-induced papillomas. In another study of DMBA-induced skin papillomas in female CD-1 mice, 2,3,7,8-T_4CDD did not act as a tumour promoter (Berry et al., 1978). Recently, Poland et al., (1982) used the hairless HRS/J strain of mouse and found that 2,3,7,8-T_4CDD promoted skin tumours initiated by DMBA or methyl-N-Nitrosoguanidine.

In other two-stage test systems, 2,3,7,8-T_4CDD was shown to act as a cocarcinogen or promoter. Kouri et al., (1978) tested the effect of 2,3,7,8-T_4CDD on 3-methylcholanthrene (3-MC) induced subcutaneous tumours in C-57 BL/6 and DBA/2 mice. 2,3,7,8-T_4CDD was cocarcinogenic in DBA/2 mice when given simultaneously with 3-MC. Pitot et al., (1980) showed 2,3,7,8-T_4CDD to be a promoter of hepatic tumours in rats initiated by diethylnitrosamine.

Clearly, whether 2,3,7,8-T_4CDD promotes or inhibits the development of previously initiated tumours appears to be highly dependent on the species, dosage, dose route, pretreatment protocol and the target organ involved.

As discussed in the previous section on genotoxicity (3.3.1), the vast majority of results suggest this compound is not genotoxic, i.e., does not act directly on chromosomes or the DNA molecule itself in such a manner as to cause chromosome breakage, mutations or DNA adducts. On this basis, the likelihood of 2,3,7,8-T_4CDD acting as an initiator is diminished. However, other short-term test results, notably the transforming ability of 2,3,7,8-T_4CDD in BHK21 cells (Hay et al., 1983), and reported mutagenic effects on L5178Y mouse-lymphoma cells (Rogers et al., 1982) indicate that the genotoxic activity and possibly tumour-initiating activity of this compound cannot be ruled out completely.

There is sufficient evidence to conclude $2,3,7,8-T_4CDD$ causes tumour production in rodents. It is difficult to conclude that $2,3,7,8-T_4CDD$ acts as a classical mutagen. $2,3,7,8-T_4CDD$ appears to produce tumours in rodents by an indirect mechanism.

3.3.3 TERATOGENICITY AND FETOTOXICITY

Teratogenicity

Although $2,3,7,8-T_4CDD$ is widely spoken of as a teratogen, there are only two teratogenic effects reported for laboratory animal tests: cleft palate in mice, and cardiovascular defects in chick embryo tests. The other effects reported are better described as embryotoxic or fetotoxic.

Teratogenic effects of $2,3,7,8-T_4CDD$ in mice were first reported in 1971 (Courtney and Moore, 1971). In the mouse embryo, $2,3,7,8-T_4CDD$ induces cleft palate and hydronephrosis. In rats, this compound appears to be more embryo- and fetotoxic than teratogenic, producing internal hemorrhages and kidney abnormalities. The key studies are tabulated in Table 3.3.3A. Cleft palate is not observed in the rats. Renal anomalies are seen in both rats and mice. $2,3,7,8-T_4CDD$ is not fetocidal in rats or mice at doses below those producing maternal toxicity. In mice the NOEL for teratogenic effects is 0.1 ug $2,3,7,8-T_4CDD$/kg/day (Smith et al; 1976). In rats, no embryo- or fetotoxic effects are observed below 0.125 ug $2,3,7,8-T_4CDD$/kg/day (Sparschu et al; 1971; Khera and Ruddick, 1973).

TABLE 3.3.3A

SUMMARY OF TERATOGENICITY AND FETOTOXICITY OF 2,3,7,8-T_4CDD IN RATS AND MICE

SPECIES, STRAIN	TERATOGENIC/ FETOTOXIC EFFECTS	DOSE (ug/kg/day)	PERIOD OF ADMINISTRATION (days of gestation)	ROUTE	REFERENCE
Mouse,					
CD-1 DBA/2J C57BL/6J	Cleft palate, kidney abnormality	1, 3	6 - 15	s.c., p.o.	Courtney and Moore, 1971
NMR1	Cleft palate	3 - 15	6-15/9-13	p.o.	Neubert etal, 1973
C57BL/6	Cleft palate, kidney abnormality	1, 3	10 - 13	p.o.	Moore etal, 1973
CF-1	Cleft palate	1, 3	6 - 15	p.o.	Smith etal, 1976
CD-1	Cleft palate, kidney abnormality	25 - 400	7 - 16	s.c., p.o.	Courtney, 1976
NMR1 C57BL	Cleft palate Cleft palate	16 25	10 - 13 11 - 13	i.p. i.p.	Hassoun and Dencker 1982
Rat,					
Sprague-Dawley	Intestinal haemorrhage	0.125 - 8	6 - 15	p.o.	Sparschu etal, 1971
CD	Kidney abnormality	0.5	6 - 15	s.c.	Courtney and Moore, 1971
Wistar	Haemorrhage (fetolethality at dose > 1ug/kg)	0.125 - 16	6 - 15	p.o.	Khera and Ruddick, 1973

In the mouse there are marked strain differences in teratogenic sensitivity to 2,3,7,8-T_4CDD. The early study of Courtney and Moore (1971) showed that inbred strains of mice were more sensitive than random bred strains. It has been postulated that the birth defects induced by 2,3,7,8-T_4CDD follow a mechanism similar to AHH induction, that is, binding to a specific cytosol receptor which translocates to the nucleus and alters gene expression. The cytosolic receptor is controlled by the A_h locus in mice. Poland and Glover (1980) found that cleft palate formation produced by 2,3,7,8-T_4CDD followed the strain distribution of the A_h locus. Strains of mice with low 2,3,7,8-T_4CDD receptor levels producing a low incidence of cleft palate when exposed to this chemical, while mouse strains with high levels of receptor developed a high incidence of cleft palate.

In five nonresponsive strains of mice (low affinity receptor) a dose of 30 ug 2,3,7,8-T_4CDD/kg/day produced a low incidence of cleft palate formation (0 to 3%) whereas the same dose produced high levels of cleft palate formation (54 to 95%) in 4 of 5 responsive strains (high affinity receptor).

Schantz et al. (1979) states that 2,3,7,8-T_4CDD has not been shown to be teratogenic in primates, although it is fetotoxic in this species at doses in the diet as low as 50ppt.

The cardiovascular teratogenic effect in chick embryos was reported by Cheung et al. in 1981. They injected varying doses of 2,3,7,8-T_4CDD from 0.009 to 77.5 picomoles (0.003- 25ng) into the egg white of chicken eggs on day zero of development and observed an increased incidence of cardiovascular malformations from about 30 to about 50 percent, with 80 percent affected at the highest dose.

Fetotoxicity

Fetotoxic effects reported include cystic kidneys (hydronephrosis), fetal gastrointestinal hemorrhage, thymic atrophy, fatty infiltration of the liver, general oedema, delayed head ossification, low birth weight, fetal resorptions, and embryolethality.

Hydronephrotic kidneys in mice exposed to 2,4,5-T containing 2,3,7,8-T_4CDD were due to retardation in fetal renal development and downgrowth of the renal papilla into the pelvis. This study by Highman et al. (1977) also demonstrated a retarded formation of fetal renal alkaline phosphatase. This result supports the hypothesis that cystic kidneys (hydronephrotic) are a fetotoxic event and not truly a teratogenic effect.

Sparschu et al. (1971) fed 2,3,7,8-T_4CDD to Sprague-Dawley rats by gavage on days 1 to 16 of gestation. The dose of 0.03 ug/kg had no adverse effects. Doses of 0.125 to 2.0ug/kg produced fetal mortality, early and late resorptions, and fetal intestinal hemorrhage with a dose-related incidence.

Murray et al. (1979) reported a three-generation study of rats exposed to 0.001, 0.01, or 0.1 ug/kg of 2,3,7,8-T_4CDD. The reproductive capacity of the rats was affected through three successive generations at dose levels of 0.01 and 0.1 ug/kg/

day, but not at 0.001ug/kg/day. Litter size at birth, gestation survival, neonatal survival and growth were decreased at the two higher levels of treatment. Fertility was decreased in the F_1 and F_2 generations but not in the first generation. No dose-related fetal abnormalities were seen in this study.

Some studies of non-human primates involving signs of frank maternal toxicity indicate a high incidence of abortion following conception (McNulty, unpublished, Allen et al; 1977, Barsotti et al; 1979). A no effect level for teratogenicity in non human primates has not been determined.

Summary

Several studies demonstrated that cleft palate and kidney defects can occur in the offspring of rats and mice treated with 2,3,7,8-T_4CDD. In mice the NOEL for teratogenicity is 0.1 ug 2,3,7,8-T_4CDD/kg/day during gestation.

3.4 HUMAN HEALTH EFFECTS OF 2,3,7,8-T_4CDD

Epidemiological studies to date do not present sufficient evidence to establish a causal association between exposure to PCDDs and the development of tumours. Since PCDDs are found as a contaminant in other chemicals of commercial importance, human exposure consists of several substances experienced simultaneously and intermittently depending on the manufacturing or application procedures. In addition, the level of PCDD contamination in the chemical products has varied over time and from place to place. This leads to difficulties in trying to ascertain the dose of PCDDs workers have been exposed to, even if the dose of the commercial product can be learned. The ascertainment is made even more difficult by attempts to reconstruct exposure using historical questionnaires.

The most noteworthy research to date has been the case-control studies carried out in Sweden by several investigators, which appear to relate soft-tissue sarcoma and malignant lymphoma to occupational exposure involving phenoxyacetic acids, chlorophenols and other chemicals with and without 2,3,7,8-T_4CDD contamination.

A review of these findings is summarized in Table 3.4A. At the Rockefeller University Symposium on Public Health Risk of the Dioxins (Lowrance, 1984), these Swedish studies were criticized on several points. One of these was the failure to provide pathological descriptions of the sarcoma. This group of tumours is quite complex and errors of naming are common. Often they are misdiagnosed and review of the cases often leads to a new diagnosis, for example, as a carcinoma rather than a sarcoma.

TABLE 3.4A

RESULTS OF CASE-CONTROL STUDIES CARRIED OUT IN SWEDEN (FROM INTERNATIONAL AGENCY FOR RESEARCH ON CANCER Vol. 30, 1983a)

Chemical Exposure	Type of Cancer and Place	Relative Risk* (95% Confidence Interval)	Reference
Phenoxyacetic acids (2,4,5-T, 2,4,-D, MCPA and T_4CDD) and/or chlorophenols	Soft tissue sarcoma Northern Sweden	5.7(2.9-11.3)	Hardell and Sandstrom (1978,1979)
Phenoxyacids alone (2,4,5-T, 2,4-D, MCPA and T_4CDD)		5.3(2.4-11.5)	
Chlorophenols alone		6.6	
DDT		1.2	
Phenoxyacetic acids and/or chlorophenols	Soft tissue sarcoma Southern Sweden	5.1(2.5-10.4)	Eriksson et al. (1981)
Phenoxyacids without T_4CDD(2,4-D, MCPA, mecoprop, dichloroprop) alone		4.2(1.3-14.5)*	
Phenoxy acids with TCDD alone		17	
All phenoxy acids alone		6.8(2.6-17.3)	
Chlorophenols alone		3.3(1.3-8.1)	
Phenoxyacetic acids (2,4,5-T, 2,4-D) chlorophenols, amitrole and picloram	Malignant lymphoma Northern Sweden	6.0(3.7-9.7)	Hardell et al. (1981)
Phenoxyacids alone		4.8(2.9-8.1)	
Chlorophenols alone (all exposures)		4.3(2.7-6.9)	
Chlorophenols alone (high exposure)		8.4(4.2-16.9)	
Chlorophenols alone (low exposure)		2.9(1.6-5.2)	
Solvents (tetrachloroethylene, trichloroethylene, benzene, styrene) (high exposure)		4.6(1.9-11.4)	
Other solvents		2.8(1.6-4.8)	
Solvents (low exposure)		4.6(1.9-11.4)	
Phenoxy acids	Colonic cancer	1.3(0.6-2.8)	Hardell (1981)
Chlorophenols		1.8(0.6-5.3)	

* Relative risk compares the cancer frequency in exposed and unexposed (or control) populations [calculated by Coggon and Acheson (1982)]

mecoprop	(2-[4-Chloro-e-methylphenoxy]propanoic acid)
dichloroprop	(2-2-[2,4-Dichlorophenoxy]propanoic acid)
picloram	(4-Amino-3,5-6-trichloropicolinic acid)

There are problems of nosology and the classification of these tumours in the International Classification of Disease manual (I.C.D.). "Soft tissue sarcoma" is a general term. "Malignant neoplasm of connective and other soft tissue" is the definition in the I.C.D. and is not synonymous with "soft tissue sarcoma". The I.C.D. definition applies only if the site of the tumour is not normal except for the extremities. A number of revisions to the I.C.D. definition over the past decade appear to have resulted in an increase in the incidence of soft tissue sarcomas reported.

There may be some inconsistency in the Swedish studies, since a cohort of Swedish railroad workers possibly exposed to the same chemical substances showed no increase in the incidence of sarcomas (Axelson et al., 1980). Similarily Hogstedt and Westerlund (1980) observed no soft tissue sarcoma in a retrospective study of Swedish forestry workers exposed to phenoxy herbicides. There have been no new Swedish cases since the reports of Hardell.

Even if the findings in the Swedish studies are real, it is still impossible to associate the tumours with a specific chemical. Relative risk of developing tumours was increased due to exposure to phenoxyacetic acids, to chlorophenols, to phenoxy acids with or without 2,3,7,8-T_4CDD, and to various solvents.

Other countries have not reported similar findings. There are negative studies from Finland (Riihimaki et al., 1983) England (May, 1982) and from New Zealand (Smith et al., 1983), but these studies may be of limited value because of small cohort size, short follow-up period and brief or low exposure to the chemicals.

TABLE 3.4B

TOXIC EFFECTS OF 2,3,7,8-T_4CDD IN MAN
(from Reggiani, 1982)

Dermatological:

- Chloracne
- Porphyria Cutanea Tarda
- Hyperpigmentation and Hirsutism

Internal:

- Liver Damage (mild fibrosis, fatty change, haemofuscin deposition and parenchymal and cell degeneration)
- Raised Serum Hepatic Enzyme Levels
- Disorder of Fat Metabolism
- Disorders of Carbohydrate Metabolism
- Cardiovascular Disorders
- Urinary Tract Disorders
- Respiratory Tract Disorders
- Pancreatic Disorders

Neurological:

- A. Peripheral
  - Polyneuropathies
  - Sensory Impairments (sight, hearing, smell, taste)
- B. Central
  - Lassitude
  - Weakness
  - Impotence
  - Loss of Libido

The above table does not include effects on the immune system which are observed in animals. Hay (1982) states that only one study showing these effects in humans has been reported to him. This report of workers exposed to 2,3,7,8-T_4CDD in 1968 had a higher proportion of cases with reduced levels of immunoglobulins IgD and IgM. However, the finding is doubted because of the poorly chosen control group.

Reggiani (1982) has also compiled the epidemiological studies relating PCDD exposure to cancer development. Table 3.4C summarizes these findings.

TABLE 3.4C

CARCINOGENICITY AND EPIDEMIOLOGY STUDIES ASSOCIATED WITH PCDDs (from Reggiani, 1982)

TYPE AND PLACE OF EXPOSURE	NUMBER OF CASES	T_4CDD LEVEL OF EXPOSURE	TIME ELAPSED SINCE EXPOSURE	FREQUENCY OF CANCER DEATH	REFERENCE
Accidental Monsanto, U.S.A.	228	Unknown	30 years	No excess	Zack 1980 Suskind
Accidental Boehringer, R.F.G.	24	Unknown	28 years	No excess	Krause 1979
Accidental BASF, R.F.G.	55	Unknown	27 years	Excess of stomach cancer	Frentzel-Beyme, 1978
Accidental Philips, Netherland	50	10,000ppb	17 years	No excess	Vos, 1978
Accidental Dow Chemical U.S.A.	60	Unknown	16 years	No excess	Cook, 1980
Accidental Spolana, Czekos.	78	24,200ppb	15 years	No excess	Jirasek, 1978
Accidental Coalite, U.K.	79	400ppb	12 years	No excess	May, 1982
Accidental Chemiewerke Linz, Austria	50	140 ppb	7 years	-	-
Herbicide Spraying, Sweden Backsprayers	348	0.00315ppt	over 10 years	Excess of stomach cancer	Axelson 1979
Herbicide Spraying, Finland Backsprayers	1960	0.00315ppt	over 10 years	No excess No lymphomas	Riihimaki, 1980
Herbicide Contamination Sweden	72	0.001ppb 0.1 ppt	over 10 years	Excess of lymphomas	Hardell, 1979

There are several other reports of human exposure to PCDDs and other chemicals, usually in an occupational setting and generally involving a short period of time, a small number of workers, or a chemical product containing a higher level of PCDDs in the past than is found in the product today. Various symptoms (e.g., chloracne, not tumours) developed following the exposure but tended to disappear slowly over several years subsequent to the incident.

Reggiani (1982) has reviewed the toxic effects of 2,3,7,8-T_4CDD in humans and Table 3.4B summarizes this review.

In general, the occupational exposure studies are inconclusive because of the lack of consistency in the findings, the small numbers exposed and the lack of knowledge concerning dose and number of the chemicals involved.

Zack and Gaffey (1983) have studied 884 chemical workers at Nitro, W. Va. Previously Zack and Suskind (1980) had reported on a study of 121 chemical workers in the same plant. This earlier study had followed an industrial accident involving trichlorophenol in which the 121 workers had developed chloracne. Of 32 deaths, analysis indicated no excess in total mortality or in deaths from malignant neoplasms.

The present study of 884 men was undertaken to increase the size of the cohort. It was pointed out that none of the potentially exposed workers had developed chloracne. Of 163 deaths, analysis also indicated no excess in total mortality or in deaths from malignant neoplasms. There was an increase in deaths from arteriosclerotic heart disease, but the authors felt this to be an artifact related to variation in the distribution of deaths from that of the U.S. Risk factors or medical care differences may exist between the plant population and the local area. The authors suggested differences may exist in smoking habits, availability and use of medical services and the specificity of diagnosis.

In humans, the most consistent toxic effect noted has been chloracne. Crow (1983) has reviewed the dermal effects of exposure to chloracnegenic chemicals. Chloracne appears to be the most sensitive indicator of poisoning in the human subject and in most cases it is a symptom of systemic poisoning. The accidental exposure to 2,4,5-trichlorophenol and its contaminants at Seveso in 1976 may be an exception. This accident resulted in dermal contact with the chemical and chloracne developed mainly in children in exposed areas of the skin. This observation is used to argue that there is no clear evidence of systemic poisoning in any of the Seveso cases.

Fara et al. (1982) have studied chloracne at Seveso and have related the geographical distribution of cases and the levels of soil contamination. This is evidence of exposure and the authors also state that no systemic toxicity was identified. They found that children with chloracne more frequently developed gastro-intestinal tract impairments than children from the same area showing no skin lesions. They were also unable to identify any major immunosuppressive effect in 48 exposed children, half of whom had developed chloracne.

Summary

The evidence for a chronic human health effect relating to carcinogenesis, coronary disease, or impairment of the immune system is not conclusive and requires further investigation. Acute exposure to PCDDs results in symptoms e.g. chloracne which slowly decrease over a prolonged period of time.

Therefore, the development of environmental standards for these chemicals should be based on animal toxicological studies.

3.5 ENVIRONMENTAL TOXICOLOGY OF PCDDs AND PCDFs

In a recent review, Kenaga and Norris (1983) have summarized much of the relevant data on the toxicity of T_4CDDs (specifically 2,3,7,8-T_4CDD) to different types of terrestrial and aquatic organisms. They also have attempted to relate the information on environmental concentrations in different media (food, soil, water, air) to the associated body or tissue residues and to pathological effects on aquatic and terrestrial organisms in a hazard assessment process.

Their treatment of the data base in this manner permits the derivation of both a no-effect dose based on laboratory exposure, either by gavage or dietary feeding trials, and an environmentally related exposure assessment based on the relationship between symptom expression, tissue or organism concentrations and the degree of contamination of the medium through which the organism is exposed.

The organization of this section follows a similar format to the Kenaga and Norris (1983) review, with the addition of any recent or omitted data. Data dealing with the concept of biological accumulation and/or bio-magnification through the food-web, excluding information on the adverse effects on the organism under study, are covered in Chapter 4 on the environmental transport, distribution, and fate of PCDDs and PCDFs.

3.5.1 TERRESTRIAL TOXICOLOGY

3.5.1.1 Toxicity to Wild and Domestic Birds

A. Laboratory Exposure

The determination of T_4CDD effects on different avian species has included oral ingestion, egg injection, and dietary feeding. In 1978, Tucker and Hudson reported the absence of any mortality 19 to 21 days after mallard ducks had ingested 0.2 ug T_4CDD/kg body weight in a 2,4,5-T formulation. Schwetz et al. (1973) also found 3-day-old Leghorn chicks showed no adverse effect from a total oral dose over a 21-day period of 2.1 ug T_4CDD/kg body weight (0.1 ug T_4CDD/kg/day). However, 80% mortality and chick oedema were evident at 21 ug T_4CDD/kg (1.0 ug T_4CDD/kg/day), while 150 ug T_4CDD/kg in 15 days (10 ug T_4CDD/kg/day) resulted in 100% mortality. These results agree with those of Greig et al. (1973), who report an LD_{50} of 25 to 50 ug T_4CDD/kg body weight for Leghorn chickens with a time to death of 12 to 21 days.

Kenaga and Norris (1983) suggest that since the main oral intake of birds is through ingestion of food, dietary feeding trials should form the basis of a realistic assessment of exposure. Accordingly, data from Kenaga (1975), Roberts and Rogers (1957), and Whitehead and Pettigrew (1972) were presented and suggest a fairly low order of toxicity from feeding trials involving T_4CDD-contaminated 2,4,5-T to mallard ducks, turkeys, and chickens. The no-observable-effect level for turkeys and chickens was in the 200 to 300 pg T_4CDD/g dietary concentration in 11- to

21-day feeding trials. In an 18-week dietary trial utilizing a concentration of 0.3 to 3 pg T_4CDD/g diet, no reproductive effects were detected in Bobwhite quail. These included parameters such as eggs laid, eggs cracked, eggs set, viable embryo, normal-hatch eggs, and 14-day old survivors. Although the no-effect level in this study was about 0.3 ng T_4CDD/kg body weight/day, levels higher than this were not utilized.

Oedema formation is the most characteristic symptom of T_4CDD poisoning in chickens (Vos 1978). Accumulation of fluids occurs in the heart sac or hydropericardium, the peritoneal cavity as well as in the lung and subcutaneous fluid (Schwetz et al., 1973). "Chick edema disease" was first encountered in the USA in 1957 and is also known as "toxic fat disease", due to the detection of toxic H_6- and T_4CDD residues in the fats of certain foods. Reduced osmotic pressure of the blood, cardiac insufficiency, and increased capillary permeability all may play causative roles in oedema formation (Vos 1978).

The general oedema-producing properties of PCDDs and extreme toxicity of 2,3,7,8-T_4CDD were detailed by Flick et al. (1972) and by Verrett (1970), who used the egg-injection technique to demonstrate that 20% embryonic mortality and 40% incidence of chick oedema were caused by 10 pg/g 2,3,7,8-T_4CDD egg concentration. Later, Schwetz et al. (1973) demonstrated that the treatment of 3-day-old chicks with 10 and 100 ug H_6CDD/kg body weight/day resulted in the development of chick oedema disease, while those fed diets containing 0.5% O_8CDD remained healthy. The dietary incorporation over 21 days of 2,3,7,8-T_4CDF at 5 ug/kg body weight/day also resulted in weight loss and pathological effects to exposed chicks (McKinney et al., 1976). Death occurred in an average of 11.5 days from the start of treatment. Reducing this rate to 1.0 ug T_4CDF/kg body weight/ day failed to achieve a no-effect status, as the test birds displayed a decrease in food consumption and body weight gain and, in some cases, inflamation and oedema in various organs.

The results of all dietary feeding or gavage trials involving the administration of T_4CDD and T_4CDF to different bird species have been summarized in Table 3.5.1.1A.

Summary

On the basis of their review data, Kenaga and Norris (1983) suggest that for birds, the semi-chronic no-effect level for T_4CDD appears to be 2.1 ug TCDD/kg body weight (0.1 ug TCDD/ kg body weight/day). The work by McKinney et al. (1976) with 2,3,7,8-T_4CDF supports this finding as visible effects were observed at 1.0 ug/kg body weight/day. No effects were observed at dietary concentrations as low as 3 pg/kg. However, the dose-response data base is insufficient to permit the derivation of a long-term dietary threshold level.

B. Environmental Exposure

Although comparatively fewer studies have been conducted to assess the effects of environmental exposure of avian species to PCDDs or PCDFs, the few studies which are available are important as they are based on Ontario data.

TABLE 3.5.1.1A
EFFECTS OF T_4CDD ON WILD AND DOMESTIC BIRD SPECIES

SPECIES	Dietary Concentration (pg/g)	TCDD INTAKE Total T_4CDD/kg body weight (ug/kg)	T_4CDD/kg b.w./day (ug/kg/day)	DURATION OF EXPOSURE (Days)	EFFECTS OBSERVED	REFERENCES
Mallard ducks		0.2			No mortality 17-19 days after treatment	Tucker and Hudson (1970)
Leghorn chicks		2.1 21 150	0.1 1.0 10.0	21 21 15	No mortality 80% mortality and chick oedema 100% mortality	Schwetz *et al.* (1973)
Leghorn chickens		25 - 50			LD_{50} (12-21 days to death)	Greig *et al.* (1973)
Chicken	500 200 100			21 21 21	90% mortality & reduced feeding and growth no mortality - reduced feeding and growth no mortality	Whitehead and Pettigrew (1972)
Turkey	259			11	No effect	Roberts and Rogers (1957)
Bobwhite quail	167 3 0.3		.003	5 126 126	50% mortality No effect on reproduction No effect on reproduction	Kenaga (1975)
Mallard ducks	278			5	10% mortality	Kenaga (1975)
Leghorn chicks			5(2,3,7,8-T_4CDF) 1(2,3,7,8-T_4CDF)	21 21	weight loss, pathological effects, death decreased food consumption, weight loss, inflamation and oedema	McKinney *et al.* (1976)

In the most detailed of these studies, the phenomenon of reproductive failure characterized by disappearance of eggs, early embryonic mortality, mortality at hatching associated with growth retardation, and congenital abnormalities among fish-eating birds on the Great Lakes, particularly Lake Ontario, has been explored (Gilbertson and Fox, 1983; Gilbertson, 1982) and related to possible contamination of the eggs with 2,3,7,8-T_4CDD. This is the only reported case of chick oedema disease, including porphyria, that has been documented in wildlife. Although 2,3,7,8-T_4CDD has now been detected and quantified in the eggs which were collected from the affected Herring gull population during the years when the disease syndrome was severe and at levels far in excess of those inducing similar symptoms in chicken embryos the authors caution that 2,3,7,8-T_4CDD may not have been the sole cause of the reproductive failure, as it is possible that additional chemicals and other pathologies were involved.

In the other Ontario study (Ryan and Pilon, 1982), PCDD and PCDF residues in liver samples from broiler chickens raised on PCP-treated wood shavings were published and related to corresponding PCDD concentrations in the wood shavings and to the pathology of the liver samples. The wood shavings were found to contain from 0.7 to 108.2 ng/g PCDD consisting of H_6CDD, H_7CDD and O_8CDD. The analysis of the affected chicken livers from these flocks revealed concentrations of H_6CDD, H_7CDD, O_8CDD, and O_8CDF ranging from 0.03 to 1.42 ng/g. Visibly unaffected livers (no gross symptoms) were (with 2 exceptions) found to be below the level of detection.

Summary

The limited amount of environmentally related PCDD exposure information prohibits the establishment of specific no-effect concentrations for avian species. The uncertainty with regard to the role of PCDDs in the Herring gull oedema syndrome also negates any use of this causitive information for dose response purposes.

3.5.1.2 Toxicity to Wild and Domestic Animals

A. Laboratory Exposure

The majority of laboratory or controlled exposures which have been conducted to assess the toxicity of PCDDs on test animals have utilized rats, mice, guinea-pigs, monkeys, and occasionally rabbits. These studies are described in greater detail in Section 3.3. The few studies involving wild or domestic animals are described below.

Cockerham et al. (1980), utilizing beachmice which had alumina dust containing 2.5 ppb T_4CDD applied to their fur 10 times in 28 days, reported that the test animals failed to suffer mortality or cellular alteration. However, significant differences in liver-to-body weight ratios were documented.

Arstila et al. (1981) studied the effects of 2,3,7,8-T_4CDD on dairy goats. The animals were fed 200 ng 2,3,7,8-T_4CDD daily for 8 weeks (5.4 ng/kg body weight/day), allowed to eliminate the contaminant for 3 weeks and then re-dosed with 400 ng/day for 4 weeks (10.8 ng/kg body weight/day). In this manner, the total two-part dose was 303 ng/kg body weight total or about 50 pg/g per day in the total diet. During the entire

13 month period, no adverse symptoms or gross autopsy results were observed in any of the test animals. Some minor effects were, however, noted in the appearance of mild fatty changes in the perilobular areas of livers from the test animals. After the second feeding, two of the animals were sacrificed and the livers were found to contain 1039 and 898 pg/g 2,3,7,8-T_4CDD, values approximately 20 times higher than the dietary 2,3,7,8-T_4CDD intake.

In similarly designed studies using beef and dairy cattle, Jensen et al. (1981) and Jensen and Hummel (1982) reported on the decline of T_4CDD contamination from the body tissues and milk, respectively. Ancillary to their main objective, assessments of animal health and, in the case of the beef cattle, tissue T_4CDD concentrations also were recorded.

In the beef cattle study (Jensen et al., 1981), animals were dosed with rations containing 24 pg/g T_4CDD for 28 days or from 0.0006 to 0.0008 ug T_4CDD/kg body weight/day for a total of about 0.02 ug/kg body weight in 4 weeks. No adverse effects were observed in any of the animals and, at the conclusion of the 4-week feeding trial, the liver tissue was found to contain an average of 10 pg/g or one-third of the dietary intake. In contrast, body fat was found to be the organ of greatest T_4CDD accumulation at an average of 84 pg/g or about 4 times the dietary T_4CDD level. In the dairy cattle study (Jensen and Hummel, 1982), the inclusion of 2,4,5-T containing T_4CDD at increasing ration concentrations of 5, 15, 50, and 150 pg/g each for 15 days and finally at 500 pg/g for 21 days for a total treatment period of 11 weeks failed to cause any noticeable symptoms or to affect milk production, body weight, or feed consumption.

Summary

It is apparent from these studies that for cattle and goats, a safe no-effect daily intake level was somewhere below 4 ug/kg body weight.

The results also serve to indicate that from a dietary ingestion aspect, it would appear that these animals would have no difficulty tolerating the even lower T_4CDD levels which would be expected from the grazing of 2,4,5-T treated range lands.

B. Environmental Exposure

A number of accidental or deliberate environmental exposures by PCDDs have provided ideal conditions for the study of the toxic effects of these chemicals through various exposure media. Unfortunately in many cases, the environmental studies which were conducted examined only the bioconcentration aspect of the problem without relating the contamination levels in the organism or the exposure medium (diet, air, water, soil) to the presence or absence of pathological effects. As already stated, these types of studies are of interest in determining the ultimate fate of the contaminants as they pass through the food chain; however, they do not address the subject of toxicity and therefore have been discussed in greater detail in Chapter 4. The two major long-term studies which have been conducted on the toxicological effects of environmental contamination by T_4CDD include the military test site for aerial spraying of Agent Orange in NW Florida and the site of the industrial accident at Seveso, Italy, in 1976. Toxic animal effects also were documented following the accidental contamination of Missouri horse arenas (Carter et al., 1975); however, due to the limited size of the contaminated areas, the short duration of exposure and the remedial actions which were taken, no long-term studies have been undertaken.

At the Eglin Air Force Base in Florida, beachmice, (Peromyscus polionotus) the most common mammalian species on the test site, were selected for a detailed research effort commencing in 1973 (Thalken and Young, 1983; Cockerham and Young, 1982). This study followed a number of more general investigations during which many mammalian, amphibian and insect species were trapped, submitted for T_4CDD analysis, and observed for gross defects, illnesses, and overall health status. Although many (about 30% of those tested) were found to be contaminated with T_4CDD, none were considered out of the "ordinary" in terms of their general, visible health status. However, tissue or organ specific pathological examinations do not appear to have been conducted on these animals.

In the case of the more detailed beachmouse study, liver tissue results ranged from 300 to 2900 pg/g T_4CDD (mean values of 1300 male and 960 female). These levels were closely related to T_4CDD levels in the soil which ranged from 10 to 1500 pg/g (mean of 326 pg/g) for the 0 to 15cm layer. Bioconcentration factors (mean liver concentrations divided by mean soil concentration) were 6 for females and 18 for males. From a pathological perspective, the examination of heart, lungs, trachea, salivary glands, thymus, liver, kidneys, stomach, pancreas, adrenals, large and small intestines, spleen, genital organs, bone, bone marrow, skin, and brain failed to reveal any significant histopathological difference between the test and control mice. This included the absence of any histopathological or ultrastructural changes in hepatic parenchymal cells (Cockerham and Young, 1982). The mean number of fetuses per pregnancy also was similar for the test and control groups. The only evidence of a statistically significant difference between the two groups was the finding that the livers of pregnant females from the test area were significantly heavier than those from the control area. As Thalken and Young (1983) point out, these studies suggest that long-term low-level exposure to T_4CDD under field conditions has had minimal effect upon the health and reproduction of the beachmouse.

Another major environmental assessment followed the accidental release of a large quantity of 2,3,7,8-T_4CDD as well as other toxic impurities including sodium hydroxide and sodium trichlorophenate from the TCP chemical plant near Seveso, Italy.

Following the accident, a number of wild and domestic animals died in the contaminated zones, with small herbivores (mainly rabbits) being the most severely affected group (Homberger et al., 1979; Fanelli et al., 1980a). The deaths, which started some days after the accident, increased noticeably within the first two weeks and decreased during subsequent months. By autumn following the July accident, virtually all of the T_4CDD had been transported from the vegetation to the upper soil layer and in Zone A, the most contaminated area, many birds, insects, snails, slugs, earthworms, lizards, mice, rats, dogs, and poultry were observed and appeared to be thriving without visible impairment (Homberger et al., 1979).

Earthworms which ingested contaminated soil accumulated 2,3,7,8-T_4CDD by a 14.5-fold factor on average (Martinucci et al., 1983). Earthworm predators, e.g. moles, have disappeared from Zone A. Other wild rodents in Zone A accumulated 2,3,7,8-T_4CDD to levels exceeding those found in the soil. No traces of 2,3,7,8-T_4CDD (level of detection 0.005 ug/kg) were found in wild rodents from uncontaminated zones.

Chemical analysis of liver samples from animals which had died or been slaughtered after the accident confirmed the presence of T_4CDD in 78 of 158 samples. In the case of rabbits, the most thoroughly studied group, liver T_4CDD concentrations ranged from 50 to 200 ug/kg in animals that had died. However, of the 14 rabbits collected in Zone A which had high concentrations in the liver (up to 300 ug/kg), all were healthy and displayed normal serological, haematological, and enzymatic values with only five showing histopathological evidence of hepatic lesions when sacrificed three months after the accident (Abbruzzi et al., 1977). As the T_4CDD was accompanied by caustic trichlorophenate which caused chemical burns in humans as well as in many of the directly exposed wildlife, it has not been possible to attribute all of the deaths immediately following the accident to T_4CDD alone (Homberger et al., 1979). The authors also indicate that many animals originally affected have recovered due to elimination of T_4CDD from the body. Liver concentrations in domestic animals slaughtered at the end of 1977, well over a year after the accident, were at least three orders of magnitude lower than in rabbits in 1976.

In another more limited ecological contamination study (Newton and Snyder, 1978), 2,4,5-T spraying on western Oregon brushfields was evaluated in terms of T_4CDD effects on Mountain beaver (*Aplodontia rufa*). After 45 days of feeding on vegetation assumed to contain between 0.22 and 2.3 pg/g T_4CDD at the instant of application, the beaver were trapped, examined, and the livers analyzed. All exposed animals were found to have histopathologically normal livers and appeared in good physical condition with no evidence of gross or microscopic lesions symptomatic of T_4CDD toxicity in liver lymphoid tissue.

Liver analyses revealed minimum detectable levels of 3 to 17 pg T_4CDD/g. These results are in agreement with a similar type of study (USEPA 1977) in which no quantifiable T_4CDD residues could be detected in the livers of livestock which had grazed on 2,4,5-T treated rangelands. Newton and Snynder (1978) point out that if one accepts a 25:1 bioconcentration ratio for liver-to-diet for no-observable effect T_4CDD concentrations (this ratio also suggested by Kenaga and Norris, 1983), these data show that the food supply for the Mountain beaver contained less than 0.125 pg/g T_4CDD, a level less than 1% of the upper doses reported as producing no effects in lifetime laboratory feeding studies (Kociba *et al.*, 1978).

Environmentally related exposure concerning the effect of pentachlorophenol-treated wooden floors on young pigs in a farrowing pen in Ontario also has been reported (Ryan 1983). After birth in a pen freshly treated with technical PCP, piglets displayed locomotor problems and vomiting, and some died within 24 to 48 hours. In contrast, the sow did not exhibit any clinical signs of toxicity. As the mortality ceased when the floor was recovered with untreated plywood, PCDD contamination was suspected. Analysis of wood, sow's milk, and various tissues from two of the affected piglets revealed very high concentrations of O_8CDD (3860 and 4600 pg/g) and H_7CDD (790 and 1285 pg/g) in the piglet skin and liver, respectively. Low or non-detectable levels of these congeners were detected in the brain, kidney, and serum samples. The wood contained 10.8, 157, and 1390 ng/g H_6-, H_7- and O_8CDD, respectively. As the PCDD tissue concentrations were in the low ng/g range with non-detectable levels of PCP being reported, PCDD contamination was concluded to be the cause of death. The analyses further pointed to dermal contact as the route of exposure, as the sow's milk was low in PCDD content.

Although the recent study by McConnell et al. (1984) does not directly address the topic of environmental exposure, it does warrant inclusion in this section on the basis of its evaluation of bioavailability of T_4CDD in soil to animals through ingestion. The animals evaluated included guinea pigs and rats who were administered (by gavage) T_4CDD contaminated soils from the Times Beach and Minker Stout (from contaminated Missouri horse arenas) sites.

The study concluded that T_4CDD in soil is biologically available to these 2 species as measured in terms of clinicopathologic syndrome in guinea pigs, hepatic enzyme induction in rats and uptake of T_4CDD in the livers of both species. The actual percentage absorption is not reported but it is indicated that even the lowest dose of T_4CDD studied (0.044 ng/kg or about 10 mg of soil per rat) induced about 4 times as much hepatic AHH activity as uncontaminated soil.

The authors also cite other studies which have demonstrated that the degree of intestinal absorption depends upon the vehicle used. In one case absorption of T_4CDD artifically added to soil was estimated to be approximately 50% less efficient than the absorption of T_4CDD in ethanol, with the efficiency decreasing as the time of T_4CDD - soil contact increased (Poiger and Schlatter, 1980). In another study (Bonaccorsi, et al. in press) using rabbits the ingestion of T_4CDD contaminated soil from the Seveso site resulted in absorption that was 68% less than that of T_4CDD in solvent.

Summary

Environmental contamination and subsequent adverse animal effects have been documented following localized, short-term accidental contamination by PCDD (Missouri Horse Arena, Ontario farrowing pen), and as a result of laboratory ingestion of T_4CDD contaminated soils by several test animals. However, no long-term toxic effects have been shown for animals living their entire life cycle in areas such as Seveso or the Eglin Air Force Base where the soil has been contaminated with T_4CDD at concentrations about 1000 times higher than would be expected following the use of a T_4CDD-contaminated product such as 2,4,5-T.

3.5.1.3 Toxicity to Natural Vegetation and Field Crops

The phytotoxicity of PCDDs and PCDFs to natural or cultivated vegetation is currently unknown. However, based on long-term ecosystem contamination studies involving the assessment of T_4CDDs on various animals, it has been concluded (Kenaga and Norris, 1983) that plants are relatively insensitive compared with animals. The one major accidental release of 2,3,7,8-T_4CDD in Seveso, Italy, was accompanied by evidence of injury to vegetation in a few species located close to the plant; however, the presence of highly caustic sodium hydroxide, and sodium chlorophenate as well as high temperatures was suggested as the likely cause of this injury. Outside this zone where 2,3,7,8-T_4CDD was confirmed in sampled vegetation, no discernable changes either to crops, shrubs, or ornamental plants were detected. Similarly, the contamination of Eglin Air Force Base with large amounts of T_4CDD present as a contaminant in Agent Orange renders any evaluation of phytotoxicity of limited value due to its co-existence with a formulated phytotoxic herbicide carrier.

In the case of aquatic vegetation, two studies dealing with the effect of T_4CDD have been published. In the first (Isensee, 1978), no effect on algae (Oedogonium cardiacum) was apparent after 32 days in water containing 1.33 ug T_4CDD/ litre. The other study also failed to detect any adverse effect on pond weeds (Elodea nuttali and Ceratophyllum emersum) after several months in water containing 0.0537 ug T_4CDD/litre.

On a cellular level, Jackson (1972) reported inhibition of mitosis and chromosomal abnormalities after treatment of a cytologic plant test system (African Blood Lily) with T_4CDD.

Summary

No firm evidence has been published to implicate any of the PCDDs or PCDFs in direct phytotoxicity to natural vegetation or field crops.

3.5.2 AQUATIC TOXICOLOGY

3.5.2.1 Toxicity to Fish

A variety of exposure and evaluation procedures have been used to assess the toxicity of PCDDs to fish. The exposures include treated food (producing LD_{50} data) and treated water (producing LC_{50} data). The response of fish eggs, various larval and juvenile stages, and adults has been examined.

Hawkes (1977) fed rainbow trout (Salmo gairdneri) food that had been treated with 2,3,7,8-T_4CDD. The food concentrations, given for 105 days, were 2.3 ng/kg, 2.3 ug/kg, and 2.3 mg/kg. The lowest treatment level produced slight liver damage but no other apparent effects. The highest treatment produced 88% mortality in 71 days. Using survival, growth, feeding activity, and fin erosion as criteria, the author estimated the no-effect level to be between 2.3ng/kg and 2.3ug/kg in the food.

In a series of papers, Helder (1980, a, b; 1981) described the effects of 2,3,7,8-T_4CDD on the early life stages of both rainbow trout (S. gairdneri) and pike (Esox lucius). In both species, a 96-hour exposure of eggs resulted in developmental and growth retardation at 0.1 ng/L.

These effects persisted for the 72-day evaluation period. Higher concentrations caused high incidences of generalized lethal oedema. Juvenile rainbow trout showed a slight growth retardation and oedema for a similar period of exposure but at 10 ng/L: Similar results were reported for pike as well. In both species, there was a short exposure period followed by an extended response period. It is also useful to note the capability of 2,3,7,8-T_4CDD to penetrate the egg membrane and affect the developing embryo.

Isensee (1979) in studying the bioaccumulation of 2,3,7,8-T_4CDD found that it was lethal to mosquito fish (Gambusia affinis) at 3 ng/L after a 14-day exposure.

Miller et al. (1973), in examining the toxicity of 2,3,7,8-T_4CDD to aquatic organisms found that for equivalent conditions of exposure, large guppies (Poecilia reticulatus) and coho salmon (Oncorhynchus kisutch) were less sensitive than smaller members of the same species. Throughout these studies, the response of the fish showed a generalized wasting syndrome. This syndrome is characterized by reduced physical activity and reduced feeding and growth. Consistently in parallel with the wasting syndrome, there is an extended period between exposure and the manifestation of the toxic reactions.

Zitko et al. (1973) fed 2,8-D_2CDF to Brook trout (*Salvelinus fontinalis*). There was no mortality in fish fed up to 122 mg/kg. The authors concluded that this particular dibenzofuran is of low acute fish toxicity.

Zitko and Choi (1973) fed fish food spiked with ug/g amounts of D_2-, T_3-, T_4- and O_8CDF to juvenile Atlantic salmon. Median mortality (time for 50% of fish to die) was 120 +/- 30 days for treated fish whereas no fish died in the control experiment. D_2-, T_3- and T_4CDF were not detected in any samples (detection limit - 0.02 ug/g wet wt). Trace amounts of O_8CDF were detected in samples from live fish. It was assumed that the toxicity was related to the T_3- and T_4CDF content of the mixture.

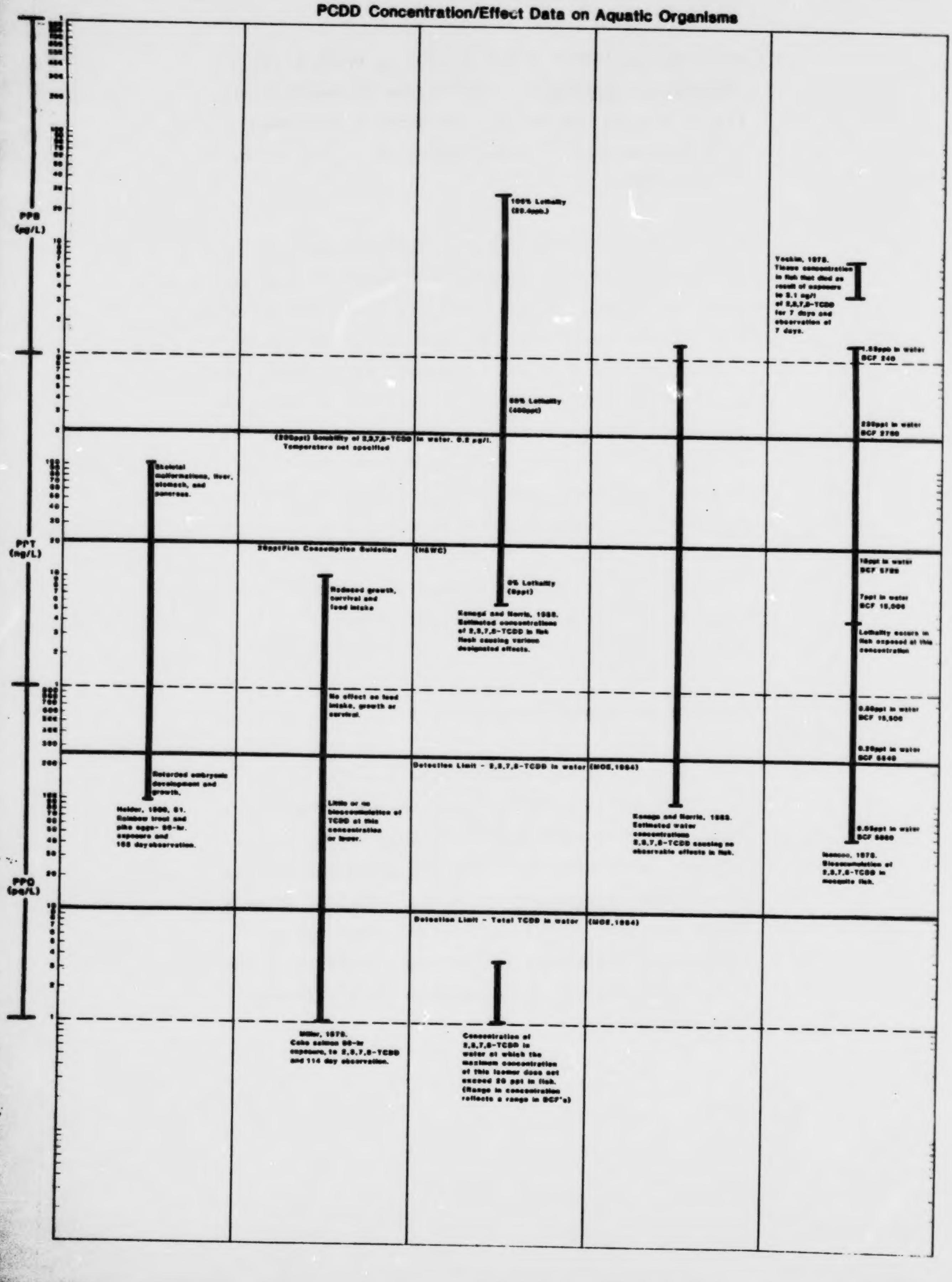
Fish toxicity and its relationship to PCDD concentration in water is summarized in Figure 3.5.2.1A.

3.5.2.2 Toxicity to Aquatic Invertebrates

Several studies have examined the toxicity of 2,3,7,8-T_4CDD to aquatic invertebrates. The animals studied were mosquito larvae (*Aedes aegypti*), pulmonate snails (*Physa sp.*), and aquatic oligochates (*Paranais sp.*) (Miller, 1973). After a 17-day exposure at 0.2 ug/L T_4CDD, there was no significant difference in the total pupation or the rate of pupation of the mosquitoes in the 30-day test period.

Figure 3.5.2.1A

PCDD Concentration/Effect Data on Aquatic Organisms



Adult snails exposed to 0.2 ug/L T_4CDD for 36 days showed no significant difference in survival when compared to the control group. T_4CDD appeared to have its major impact on the reproductive success of the snails, i.e., on the growth and survival of the juvenile life stages.

Isensee (1978) exposed Daphnia magna and Physa sp. to T_4CDD concentrations up to 1.33 ug/L for 1 to 30 days. There were no apparent adverse effects under these exposure conditions. Aquatic oligochaete worms (Paranais sp.) were exposed to 0.2 ug/L for 55 days. Assessment of the population at the end of the exposure period showed that T_4CDD exerted its principal effect on reproduction rather than growth of the individuals.

Summary

In summary, it can be seen that fish exposed to 2,3,7,8-T_4CDD in water react to very low concentrations (about 1 ng/L) and usually with an extended latency period. Embryonic and larval forms of fish appear to be most sensitive. Studies on aquatic fauna other than fish are very limited.

3.6 COMPARATIVE TOXICOLOGY OF PCDDs AND PCDFs

The physical and chemical properties of PCDDs and PCDFs are described in Sections 2.1.1 annd 2.1.2.

3.6.1 COMPARATIVE TOXICITY OF PCDD CONGENERS

While there is insufficient information at present to assess the toxicological properties of all PCDD congeners, comparative toxicology of some isomers has been studied. Although the present information is far from complete, some general toxicological ranking of isomers may be made.

3.6.2 ACUTE TOXICITY OF PCDDs

Acute toxicity studies have assessed the PCDDs in terms of LD_{50}. Congeners with low LD_{50} numbers are the most potently toxic. The acute toxicities of various PCDD isomers, measured in two mammalian systems, guinea pig and mouse, have been summarized by Kociba and Schwetz, (1982a). These data are shown in Table 3.6.2A.

The acute toxicity of 2,3,7,8-T_4CDD is summarized in Section 3.2.2.

Other PCDD congeners, however, are generally less potent than 2,3,7,8-T_4CDD. Toxicities can vary by factors of 100 to 10,000-fold for closely related isomers such as 2,3,7,8- and 1,2,3,8-T_4CDD and 1,2,3,7,8- and 1,2,4,7,8-P_5CDD. Structure-activity considerations and limited LD_{50} data (Table 3.6.2A) indicate the more toxic members of each chlorine substitution group substituted in the 2,3,7 and 8 positions.

Within this highly toxic group, 1,2,3,7,8-P_5CDD is only slightly less toxic than 2,3,7,8-T_4CDD in both rodent species. The hexachlorinated-isomers, 1,2,3,4,7,8-, 1,2,3,6,7,8- and 1,2,3,7,8,9- H_6CDD, are 50-fold less potent than 2,3,7,8-T_4CDD.

1,2,4,7,8-P_5CDD and 1,2,3,4,6,7,8-H_7CDD are 100 to 1000-fold less toxic, and 2,3,7-T_3CDD and 2,8-D_2CDD are 15,000-and 150,000-fold less potent than 2,3,7,8-T_4CDD in guinea pig. Mice, however, are comparatively more sensitive to 2,3,7-T_3CDD. O_8CDD, although not tested in these species, has been reported to have a LD_{50} greater than 1gm/kg in rats and mice (Schwetz *et al.*, 1973).

TABLE 3.6.2A

ACUTE TOXICITIES AND AHH INDUCTION EFFICIENCIES OF PCDD

		ACUTE TOXICITY		AHH INDUCTION			
				in vivo			in vitro
		Guinea Pig LD_{50} ug/kg	Mouse LD_{50} ug/kg	Chicken Embryo ED_{50} ug/egg	C57BL/6 Mouse ED_{50} ug/kg	DBA/2B Mouse ED_{50} ug/kg	Rat Hepatoma EC_{50} ug/L
DD			>50,000	-			-
M_1CDD				-			-
D_2CDD	1,3			-			-
	1,6			-			-
	2,3			-			-
	2,7		>2,000,000	-			-
	2,8	>300,000	847,000,000	6.3			-
T_3CDD	1,2,4			1.4			
	2,3,7	29,444	>3,000	1.1			330
T_4CDD	1,2,3,4						-
	1,2,3,8						201
	1,3,6,8						-
	1,3,7,8			1.3			29
	2,3,7,8	2	283.7	0.007	0.27	3.9	0.12
	1,3,7,9						
P_5CDD	1,2,3,7,8	3.1	337.5				2.0
	1,2,4,7,8	1,125	>5,000	0.012			45,000
H_6CDD	1,2,3,4,7,8	72.5	825	0.16			3.0
	1,2,3,6,7,8	70-100	1,250	0.55			13
	1,2,3,6,7,9						32,000
	1,2,3,7,8,9	60-100	>1,440	0.01			18
	1,2,4,6,7,9			-			-
H_7CDD	1,2,3,4,6,7,8	>600		1.61			59
	1,2,3,4,6,7,9						1,600
O_8CDD	1,2,3,4,5,6,7,8		>4,000,000				9,200

Adapted from Kociba and Schwetz, 1982a.

- no induction at dose tested.

3.6.3 BIOCHEMICAL RESPONSES OF PCDDs

As mentioned in Sections 3.2.1 and 3.2.4.1, 2,3,7,8-T_4CDD is a potent inducer of several drug-metabolizing enzymes, particularly AHH. Within a series of structurally diverse PCDDs, there was an excellent correlation between their rank order AHH induction potencies in several systems (chick embryo, C57BL/6 mice and DBA/2 mice) and their toxicities (LD_{50}) (illustrated in Table 3.6.2A). A summary of the potency of various PCDD isomers, expressed as ED_{50} (dose required to induce 50% of the maximum level of these enzymes) is shown in Table 3.6.2A. This response is summarized for three in vivo systems (chick embryo, C57BL mice, and DBA mice). DBA/2B mice have been included in this summary although they are insensitive to 3-methylcholanthrene (3MCA), an inducer of these P_{448} enzymes in C57BL mice. 2,3,7,8-T_4CDD does induce AHH activity in this non-responsive DBA/2 strain of mouse albeit at higher concentrations than for the responsive C57BL mice. Also included in this summary is the effect of PCDD on the in vitro induction of these enzymes in rat hepatoma tissue culture.

The PCDD isomer 2,3,7,8-T_4CDD, the most acutely toxic isomer, is also the most effective inducer of AHH (Table 3.6.2A). The chick embryo system is the most sensitive with comparable sensitivity to the in vitro system. Mice are slightly less sensitive, with the 3-MCA "uninduceable" strain DBA/2B requiring more than 10 fold greater levels to achieve 50% of their maximum AHH levels.

The comparative AHH induction potency of various PCDD isomers parallels the acute toxicity of these isomers (Poland et al., 1979; Bradlaw and

Casterline, 1979). Evaluation in the rat hepatoma system is the most informative (Bradlaw and Casterline, 1979) since a larger number of isomers have been tested. Relative to the potency of 2,3,7,8-T_4CDD, two isomers 1,2,3,7,8-P_5CDD and 1,2,3,4,7,8-H_6CDD are approximately 1/10 as potent. Four other isomers are 1/100-1/500 as potent while six additional isomers have potencies 1/1000 to approximately 1/100,000 that of 2,3,7,8-T_4CDD. Ten of the PCDD isomers were inactive at doses up to 50,000 pM/plate (4 mL).

The pivotal role of the cytosolic receptor protein in regulating AHH induction (Okey et al., 1979) is reflected in differing binding avidities of various PCDDs, PCDFs and PCDD and PCDF mixtures to this molecule (Poland et al., 1976; Hutzinger et al., 1981; Sawyer et al., 1983 and Bandiera et al., 1984). While the relative binding avidities of 26 PCDFs have recently been published (Bandiera et al., 1984), relative binding avidities for only about 10 PCDDs have been published (Poland et al., 1976). Excellent correlation between rank order AHH induction potencies, LD_{50} values and receptor binding avidities exist for the PCDDs and PCDFs examined so far.

Summary

The comparative studies of PCDDs in terms of potency of AHH induction have provided valuable insights into the comparative toxicity of these compounds. Although microsomal monooxygenase activity is not suspected to be the cause of PCDD toxicity, the correlation between relative potencies of acute toxicity of PCDDs and AHH induction potency enables some generalization regarding comparative toxicities of these compounds to be made.

As a result of these assays, as well as acute toxicity studies, it has been demonstrated that several other PCDDs, in addition to 2,3,7,8-T_4CDD have relatively potent toxic properties. In addition to those congeners already evaluated, the screening of other PCDD congeners for potential acute toxicity is practical using these assays. Lastly, these studies have enabled postulation of structural-functional relationships of various PCDDs.

3.6.4 TERATOLOGY OF PCDDs

Relatively few PCDD congeners have been evaluated for their potency in inducing teratological or fetotoxic responses. However, a range of congeners have been studied and data from these studies have been summarized by Kociba and Schwetz (1982). A summary of these data is presented in Table 3.6.4A.

Of the isomers studied, 2,3,7,8-T_4CDD is the most potently fetotoxic, inducing this response in rats at doses between 0.125 and 0.25 ug/kg/day. No fetotoxic effects were reported for doses less than 0.125 ug/kg/day.

Two congeners, D_2CDD and H_6CDD also produced teratological effects. A mixture of H_6CDD was active at doses in the range of 1-10 ug/Kg/day. The isomer 2,7-D_2CDD was apparently teratogenic at doses of 1,000-2,000 ug/kg/day. Three other isomers were found to be non-teratogenic in the rats.

A study of other members of the PCDDs by Courtney (1976) showed that they were relatively non-toxic and were non-teratogenic at the doses studied. A mixture of D_2CDD and T_3CDD produced a slight increase in mild hydronephrosis in mouse pups from one litter but no malformations at a higher dose. The 1,2,3,4-T_4CDD compound did not increase the incidence of malformations at any dose level. Oral administration of 5 or 20 mg/kg/day of O_8CDD to pregnant mice did not alter fetal development.

Summary

The information available to date is insufficient and considerably more isomers must be tested before the comparative teratogenicity of PCDDs can be evaluated.

TABLE 3.6.4A

COMPARATIVE TERATOLOGY OF PCDD ISOMERS IN RATS[1]

ANALOGUE	ISOMER	DOSE (ug/kg/day)	EFFECT	REF.
M_1CDD	2	1,000-2,000	N.F.A.	(2)
D_2CDD	2,3	1,000-2,000	N.F.A.	(2)
	2,7	250-500	N.F.A.	(2)
		1,000-2,000	Slight myocardial oedema	(2)
		100,000	N.F.A.	(4)
T_4CDD	1,2,3,4	50-800	N.F.A.	(2)
	2,3,7,8	0.03	N.F.A.	(3)
		0.125	N.F.A.	(2)
		>0.125	Weight loss, oedema, haemorrhage embryolethal	(3)
		0.25-2	Fetal oedema, haemorrhage	(2)
		>4	Embryolethal Dam toxicity	(2)
H_6CDD	Mixture	0.1	N.F.A.	(4)
		1-10	Subcutaneous oedema	(4)
		100	Fetotoxic, fetal anomalies	(4)
O_8CDD	1,2,3,4,5,6,7,8	100,000	N.F.A.	(4)
		500,000	N.F.A.	(4)

1 - From Kociba and Schwetz, 1982b.
2 - Khera and Ruddick, 1973
3 - Sparschu et al., 1971
4 - Schwetz et al., 1973
N.F.A. - No fetal anomalies.

TABLE 3.6.5A

SUMMARY OF DATA FROM ANIMAL CARCINOGENICITY BIOASSAYS OF PCDDs[1]

ANALOGUE	ISOMER	RAT		MOUSE	
		Dose	Effect	Dose	Effect
DD		5,000 ppm in diet 10,000 ppm in diet	NC NC	5,000 ppm in diet 10,000 ppm in diet	NC NC
D_2CDD	2,7	5,000 ppm in diet 10,000 ppm in diet	NC NC	5,000 ppm in diet 10,000 ppm in diet	NC Suggestive carcinogenic response
T_4CDD	2,3,7,8	0.001 ug/kg/day 0.007-0.01 ug/kg/day	NC Carcinogenic response	0.03 ug/kg/day 0.07 ug/kg/day	NC Carcinogenic response
H_6CDD	Mixture 1,2,3,6,7,8 1,2,3,7,8,9	1.25 ug/kg/day 2.5 ug/kg/day 10.0 ug/kg/day	Carcinogenic response at higher dose	2.5 ug/kg/day 5.0 ug/kg/day 10.0 ug/kg/day	Carcinogenic response at higher dose

[1] - From Kociba and Schwetz, 1982b.

NC - No carcinogenic response.

3.6.5 CARCINOGENICITY OF PCDDs

Only 2 other PCDDs, other than 2,3,7,8-T_4CDD, have been studied in mammalian carcinogenicity bioassays.

2,3,7,8-T_4CDD induced tumours in rats at doses of 0.007-0.01 ug/kg/day and induced tumours in mice at 0.07 ug/kg/day (Kociba and Schwetz, 1982) (Section 3.6.3). A mixture of H_6CDD isomers has also been shown to be carcinogenic in rats and mice, inducing tumours in the range of 2.5-10 ug/kg/day (NCI, 1980). In addition, 2,7-D_2CDD, at a dose of 10,000 mg/kg in feed, induced a "suggestive" carcinogenic response in mice (Kociba and Schwetz, 1982b). Table 3.6.5A contains a summary of these data.

Summary

The available information from carcinogenicity bioassays of PCDD congeners other than 2,3,7,8-T_4CDD is at present insufficient to assess the comparative carcinogenicity of these compounds.

3.6.6 COMPARATIVE TOXICITY OF PCDFs

PCDFs, have not as yet been thoroughly evaluated for toxicological potencies. A recent study has evaluated the structure-activity relationships of 26 PCDFs in terms of binding avidity to the 2,3,7,8-T_4CDD cytosolic receptor protein and induction of AHH and EROD in rat hepatoma cell cultures. These *in vitro* studies were complimented with *in vivo* studies of the relative activities of selected PCDFs on induction of hepatic microsomal monoxygenase, body weight gain and thymus weights in immature male Wistar rats (Bandiera *et al.*, 1984). This information and some earlier studies (cited below), though limited, provide some means to evaluate the comparative toxicity or biological activity of these compounds. No carcinogenicity studies of PCDFs are available.

Acute Toxicity

Studies of the acute toxicity of PCDFs are limited to testing 3 PCDFs on guinea pigs, mice and rhesus monkeys (Moore et al., 1979). Symptoms of acute toxicity are similar to those caused by PCDDs. As in the case of PCDDs, guinea pigs are the most sensitive species. LD_{50} values for this species are as follows: 7ug 2,3,7,8-T_4CDF/kg, <10 ug 2,3,4,7,8-P_5CDF/kg and 120ug 2,3,4,6,7,8-H_6CDF/kg (McKinney and McConnell, 1982). As with PCDDs, average time to death was 12 - 16 days.

Estimated LD_{50} values for 2,3,7,8-T_4CDF for other species are as follows: >6000 ug/kg (mouse), >1000 ug/kg (rat); 300 - 1000 ug/kg (monkey) (Moore et al., 1979 McNulty et al., 1982).

Pretreatment of immature male Wistar rats with 500 or 1,000ug/kg of 2,3,7,8-T_4CDF or 2,3,4,7,8-P_5CDF caused significant body weight loss, thymic atrophy and induction of hepatic cytochrome P-448 dependent monooxygenases (Bandiera et al., 1984). PCDF congeners which contained less than 4 lateral chlorine substitutents (2,3,4,8- and 1,2,4,8-T_4CDF and 1,2,4,7,8- and 1,2,4,7,9-P_5CDF) did not cause body weight loss or thymic atrophy at these dose levels.

Enzyme Induction

Early work of Poland and Glover (1973) showed that PCDDs that were most potent in inducing AHH in the chicken embryo were also the most toxic. This relationship was later extended to PCDFs using the chicken embryo system (Poland et al., 1976).

TABLE 3.6.6A

HEPATIC ARYL HYDROCARBON HYDROXYLASE INDUCTION EFFICIENCY BY PCDF ISOMERS

		in vivo Level of AHH activity relative to control				*in vitro* Calculated EC_{50}	
		rat liver		mouse liver		rat hepatoma cells	
ANALOGUE	ISOMER	a)	b)	c)	d)	e) (ug/plate)	f) (ug/flask)
DF						-(8.4)	
M_1CDF	4						2035
D_2CDF	2,6					-(11.9)	14620
	2,8	-(10,000)					9360
	2,3						519
T_3CDF	1,3,6						687
	2,4,6					126.5(P)	
	1,3,8						5267
	2,3,4						41
	2,4,8		-(5)				
	2,6,7						760
	2,3,8*					0.8	676
T_4CDF	1,4,6,8					-(16.1)	
	1,2,4,8						3672
	1,3,6,7	-(5,000)	-(5)				
	1,3,6,8	-(5,000)					
	2,3,4,6						404
	2,4,6,8					-(16.1)	
	1,2,7,8*	5(1,000)					
	2,3,4,8*						1217
	2,3,6,7*	5(5,000)	-(5)	0.8(30)	1.3(30)		
	2,3,6,8		1.3(5)			20.8(P)	316
	2,3,7,8*	11(5,000)	4(5)**	1.0(30)	13.6(30)**	0.004	1.2
P_5CDF	1,2,4,6,7						111
	1,2,4,6,8	-(5,000)					
	1,2,4,7,9						12.8
	1,2,3,4,8*						71
	1,2,3,6,7*		-(5)	0.8(30)	1.0(30)		
	1,2,4,7,8		1.4(5)			1.5	36
	1,2,6,7,8*	6(1,000)	-(5)				
	1,3,4,7,8					0.09	
	2,3,4,6,7*	12(1,000)	-(5)	0.8(30)	2.7(30)		
	1,2,3,7,8*	7(10,000)	-(5)	1.1(30)	6.9(30)**		0.9
	2,3,4,7,8*	9(1,000)	2.0(5)**	0.9(30)	7.6(30)**		0.09
H_6CDF	1,2,3,4,6,7*	6(1,000)	-(5)	0.9(30)	1.2(30)		
	1,2,4,6,7,8						15.9
	1,2,3,4,7,8*	8(10,000)	1.3(5)	0.9(30)	1.6(30)		0.13
	1,2,3,6,7,8*		-(5)				0.55
	2,3,4,6,7,8*						0.26
H_7CDF	1,2,3,4,6, 8,9					1.4	
T_4CDD	2,3,7,8*		7.0(5)	10.9(30)	21.0(30)	0.0005	0.002

a) Yoshihara et al., 1981 (Wistar Rat - male - liver)
b) Nagayama et al., 1983a (Wistar Rat - male - liver)
c) Nagayama et al., 1983b (DDD Mouse - liver)
d) Nagayama et al., 1983b (AKR Mouse - liver)
e) Bradlaw and Casterline, 1979 (Rat Hepatoma Cell Culture)
f) Bandiera et al., 1984 (Rat Hepatoma Cell Culture)
- indicates no induction at dose tested, (ug/kg body weight or ug/container)
* chlorine substitution in 3 lateral positions (2, 3, 7 or 8) and one vicinal hydrogen or chlorine substitution in all 4 lateral positions
** indicates significant induction ($p < 0.01$)
(P) indicates projected EC_{50}

Subsequent studies have reinforced this relationship between AHH induction and toxicity using mice, rats or rat cell cultures (Bradlaw and Casterline, 1979; Yoshihara et al., 1981, Nagayama et al., 1983a, Nagayama et al., 1983b and Bandiera et al., 1984) as shown in Table 3.6.6A.

In this table, two expressions of results are presented. For those studies in mice or male Wistar rats, higher levels of AHH activity at lower doses of isomer are indicative of greater potency. For those studies on the rat hepatoma tissue cell system, lower EC_{50} values reflect higher potency.

Teratology

Only one very recent report has directly investigated the teratogenicity of PCDFs in mammals (Weber et al., 1984). Pregnant C57BL/6N mice were treated with 0, 250, 500 or 1,000ug 2,3,7,8-T_4CDF/kg on day 10 of gestation or with 0, 10, 30, 50, or 100ug 2,3,7,8-T_4CDF/kg on gestation days 10 to 13. Dose-related increases in cleft palate and hydronephrotic kidney were observed in the offspring. Kidney changes were the most sensitive endpoint and were observed in all the litters of dams receiving either a single dose of 100ug/kg on gestation day 10 or 4 daily doses of 30ug/kg on days 10 to 13 of gestation. Treatments were not toxic to the dams. The maternal LD_{50} is estimated to be 6 to 10mg/kg. A previous study has indicated that PCDFs are transferred to fetuses and offspring in mice (Nagayama et al., 1980).

Mutagenicity

Schoeny (1982) tested dibenzofuran and 4 PCDF congeners for mutagenicity using the Ames test

(*Salmonella typhimurium* strains TA 98 and TA 100 with and without microsomal activation). Dibenzofuran, 2,9-D_2CDF, 3,8-D_2CDF, 2,3,7,8-T_4CDF and O_8CDF were not mutagenic for any *Salmonella* strain over the concentration range tested.

More recently Mortelmans *et al.*, (1984) did not observe mutagenicity when dibenzofuran was tested with *S. typhimurium* strains TA 98, TA 100, TA 1535 and TA 1537 with or without microsomal activation.

Summary

Animal (LD_{50}, body weight gain, thymus/body weight ratio, teratology) and *in vitro* (enzyme induction, cytosolic receptor binding) studies indicate that there are pronounced differences in the toxic and biological effects of the different PCDD and PCDF congeners. These differences are strongly correlated with the chlorine substitution patterns of the PCDD or PCDF molecule. The isomers with the highest activity and acute toxicity are those with 4 to 6 chlorines and all lateral (2,3,7 and 8) positions substituted with chlorine. Further chlorine substitution or removal of lateral chlorines results in congeners with decreased toxicities or biological activity. Based on this structure-activity relationship and assuming that 3 or 4 of the 2,3,7 or 8 positions are substituted with chlorine the number of potentially toxic isomers in each chlorine substitution group may be estimated.

Using this approach 23 of the 75 PCDD and 40 of the 135 PCDF congeners are potentially toxic. Using the criteria of chlorine substitutions in all 4 lateral positions and data from LD_{50} and enzyme induction studies 5 PCDDs (2,3,7,8-T_4CDD,

1,2,3,7,8-P_5CDD, 1,2,3,4,7,8-H_6CDD, 1,2,3,7,8,9-H_6CDD and 1,2,3,6,7,8-H_6CDD) and 7 PCDFs (2,3,7,8-T_4CDF, 1,2,3,7,8-P_5CDF, 2,3,4,7,8-P_5CDF, 1,2,3,6,7,8-H_6CDF, 1,2,3,7,8,9-H_6CDF, 1,2,3,4,7,8-H_6CDF and 2,3,4,6,7,8-H_6CDF) can be identified as extremely toxic (guinea pig LD_{50} < 500 ug/kg).

Toxicities or biological activities of PCDDs or PCDFs outside these two extremely toxic and potentially toxic categories drop rapidly with M_1-, D_2- and O_8- substituted congeners showing little or no toxicity.

In the absence of detailed analysis of the chemical and toxicological properties of the wide variety of PCDDs and PCDFs in environmental samples the remarkable rank order correlations of AHH induction, receptor binding avidity and toxicity with chlorine substitution patterns suggests the great potential of these tests for biological monitoring (Section 5.3.2) and hazard assessment (Section 5.3.3).

2,3,7,8-T_4CDD is the most toxic and best understood PCDD and the proposed standard for PCDDs and PCDFs (Section 3.7) is based on this congener. Other PCDDs and PCDFs must therefore be prorated so that "2,3,7,8-T_4CDD" toxic equivalents can be derived.

Chronic toxicity studies exist for only 4 PCDDs. Consequently the only practical approach to prorating chronic effects of PCDDs and PCDFs must be based on current knowledge of structural correlations derived from short term experiments (Section 3.6.7).

3.6.7 2,3,7,8-T_4CDD TOXIC EQUIVALENTS

Available information on toxicity centres around 2,3,7,8-T_4CDD. This is the main contaminant in 2,4,5-T_3CP related products and wastes. It represents a small amount of the PCDD present in P_5CP-related products and wastes where much larger amounts of H_6CDDs, H_7CDDs and O_8CDDs predominate. Fly ash, combustion sources, PCBs and PCB fires on the other hand contain or emit a complex mixture of PCDDs and PCDF.

Hence, assessing the net toxicity of complex mixtures is a problem and various approaches have been used to deal with this uncertainty as depicted in Table 3.6.7A. The approaches divide into three general areas:

(a) Assumptions and associated scaling factors regarding the toxicity of congeners relative to 2,3,7,8-T_4CDD; assumptions are based on relative enzyme inducing abilities (i.e., Example 2).

(b) Assumptions and associated scaling factors regarding the toxicity of congeners relative to 2,3,7,8-T_4CDD; assumptions are based on the structure-activity hypothesis of Poland or on prudence (i.e., Examples 5 and 1, respectively).

(c) Biological monitoring methods where the activity of the mixture is compared to 2,3,7,8-T_4CDD (i.e. Examples 3 and 4).

TABLE 3.6.7A
2,3,7,8-T_4CDD TOXIC EQUIVALENTS

E.G. NO.	SAMPLE	ANALYTICAL METHOD QUANTITATED	ASSUMPTIONS OR BIOANALYTICAL METHOD USED	X-FOLD TOXICITY OF THE MIXTURE ABOVE THE MEASURED VALUE FOR 2,3,7,8-T_4CDD	REFERENCE
1	Particulate emission from Resource Recovery Facility	Total T_4CDD's and 2,3,7,8-T_4CDD as 7% of total T_4CDDs	Assumed all T_4CDD's as toxic as 2,3,7,8-T_4CDD.	14X	(Redford, 1981)
2	Flue gas emission from municipal incinerator	2,3,7,8-T_4CDD & Isomer Groups T_4CDD T_4CDF P_5CDD P_5CDF H_6CDD H_6CDF H_7CDD H_7CDF O_8CDD O_8CDF	ASSUMED ENZYME INDUCTION ABILITY Assumed relative toxicity based on following microsomal enzyme induction ability: 2,3,7,8-T_4CDD 1.0 Other T_4CDD 0.01 T_4CDF 0.1 P_5CDD 0.1 P_5CDF 0.1 H_6CDD 0.1 H_6CDF 0.1 H_7CDD 0.01 H_7CDF 0.01 O_8CDD - O_8CDF - Assume entire isomer group has the same induction ability as the most potent inducer in that group (Conservative). Used above factors as weighting factors to convert analytically measured isomer group masses to 2,3,7,8-T_4CDD equivalents.	60X	(Swiss Federal Office for Environmental Protection, 1982)
3	Fly ash extract	2,3,7,8-T_4CDD	RECEPTOR BIOASSAY METHOD Measured dose-dependant ability of extract to displace radiolabelled 2,3,7,8-T_4CDD from the cytosol receptor protein; EC_{50} values i.e., the amount of unlabelled ligand to displace one-half of the [3H]-T_4CDD are used for comparing mixtures to 2,3,7,8-T_4CDD.	45X	(Hutzinger, *et al.*, 1981)

TABLE 3.6.7A (Cont'd)

<table>
<tr><th>E.G. NO.</th><th>SAMPLE</th><th>ANALYTICAL METHOD QUANTITATED</th><th>ASSUMPTIONS OR BIOANALYTICAL METHOD USED</th><th>X-FOLD TOXICITY OF THE MIXTURE ABOVE THE MEASURED VALUE FOR 2,3,7,8-T_4CDD</th><th>REFERENCE</th></tr>
<tr><td>4</td><td>Fly ash extract</td><td>2,3,7,8-T_4CDD</td><td>ENZYME INDUCTION METHOD
Measured dose-dependent ability of extract to induce the aryl hydrocarbon hydroxylase (AHH) enzyme system. EC_{50} values (i.e. the amount of half-maximal induction) are measured by two techniques (i.e. AHH & EROD) and are used for comparing mixtures to 2,3,7,8-T_4CDD.</td><td>8X (AHH)
26X (EROD)</td><td>(Sawyer, et al., 1983)</td></tr>
<tr><td>5</td><td>Flue gas emission from municipal reactor</td><td>2,3,7,8-T_4CDD & Total Homologues
T_4CDD T_4CDF
P_5CDD P_5CDF
H_6CDD H_6CDF
H_7CDD H_7CDF</td><td>ASSUMPTION OF POLANDS'S HYPOTHESIS re Toxicity
Assume toxicity of PCDD's and PCDF's related to substitution in three of four of the 2,3,7 and 8 positions.
These congeners were considered to be of concern; the rest were ignored:
i.e.
<table>
<tr><th></th><th>Total # of isomers</th><th># of Toxic isomers</th></tr>
<tr><td>T_4CDD</td><td>22</td><td>5 (incl. 2,3,7,8-T_4CDD)</td></tr>
<tr><td>P_5CDD</td><td>14</td><td>7</td></tr>
<tr><td>H_6CDD</td><td>10</td><td>7</td></tr>
<tr><td>H_7CDD</td><td>2</td><td>1</td></tr>
<tr><td>T_4CDF</td><td>38</td><td>8</td></tr>
<tr><td>P_5CDF</td><td>28</td><td>14</td></tr>
<tr><td>H_6CDF</td><td>16</td><td>12</td></tr>
<tr><td>H_7CDF</td><td>4</td><td>2</td></tr>
</table>
Further assume:
- 2,3,7,8-T_4CDD = 3.3% of total T_4CDD.
- All "Toxic" compounds have a toxicity of 10% compared to 2,3,7,8-T_4CDD.
- Isomers are equally distributed in each isomer group.
Calculation:
$$\left[\text{2,3,7,8-}T_4\text{CDD toxic equivalent of an isomer group}\right] = \frac{\left[\text{Analytically measured mass of isomer group}\right]}{\left[\text{\# isomers in group}\right]} \times \left[\text{\# of "toxic" isomers}\right] \times 0.1$$</td><td>80X</td><td>(Hutzinger, et al., 1983)
(Grant, 1977)</td></tr>
</table>

Examples 2 and 5 require analytical quantitation of congeners and may require quantitation of 2,3,7,8-T_4CDD. But in many cases (e.g. flue gas emissions, fly ash) there is sufficient world-wide data that a worst-case assumption regarding the amount of 2,3,7,8-T_4CDD in total T_4CDD's can be made.

Examples 3 and 4 are much simpler and less costly to apply since only the extract of the appropriate matrix is required. However, some of these biological methods have validation problems. It was noted recently that quantitative differences (about 30-fold) between the induction and receptor binding methods are not understood (Sawyer et al., 1983). Non-specific binding of 2,3,7,8-T_4CDD to the receptor protein yields higher apparent activities using this assay.

Estimation of gross toxicity of mixtures is possible using analytical as well as bioanalytical methods. The simplicity, low cost and sensitivity (especially systems based on cell cultures such as AHH induction and hyperkeratinization as noted in section 6.2) of these latter methods make them attractive tools for quick monitoring as well as for the enforcement of standards.

The proposed maximum allowable daily intake of PCDDs and PCDFs (from all routes of exposure) is derived in section 3.7. Options for prorating the proposed standard are outlined in Table 3.6.7B.

Options 1) and 2) are unrealistic or not applicable to the Ontario situation. Options 4) and 6) require a sophistication of analysis and validation not yet attainable in Ontario. Option 3) has great practical merit and is a simple form of Option 5).

It is recommended that Option 5) shows the greatest promise for prorating PCDD and PCDF toxicity at the present time. It is also practical, since in some cases, analytically measured masses of isomer groups are available. A proposed prorating formula, its assumptions and its application to worst case examples from Ontario data is set out below.

Proposed Numerical Relationship for Converting Isomer Group Residue Data to 2,3,7,8-T_4CDD Equivalents

Chronic toxicity studies only exist for 4 PCDDs (Section 3.6.5). On the other hand, acute toxicity and biological activity (enzyme induction, cytosolic receptor binding) studies are more numerous.

Evidence of a common mechanism of action for PCDDs and PCDFs can be deduced from the strong structure-activity and structure-toxicity relationships revealed in these short term studies. Consequently the only practical approach to prorating chronic effects of PCDDs and PCDFs must be based on current knowledge of structural correlations derived from short term experiments.

It is assumed that the magnitude of the toxic or biological responses induced by the various PCDDs and PCDFs is dependent on the number and substitution of chlorine atoms on the congener molecule. 2,3,7,8-substituted congeners are most likely to be biologically active and persist in

TABLE 3.6.7B
OPTIONS FOR PRORATING THE PROPOSED STANDARD

OPTION	PRO	CON
(1) all congeners considered equitoxic to 2,3,7,8-T_4CDD	- very large implicit safety factor	- not supported by scientific evidence - standard easily exceeded
(2) apply to 2,3,7,8-T_4CDD alone	- well documented - MOE has analytical capability - standard easily attainable	- does not address other toxic congeners - not relevant to Ontario situation (2,3,7,8-T_4CDD absent or in trace amounts in most media)
(3) prorate or ignore O_8CDD and O_8CDF and consider all other congener equitoxic to 2,3,7,8-T_4CDD	- retains large safety factor - addresses the known low toxicity of O_8CDD and O_8CDF	- ignores other congeners of low toxicity
(4) divide congeners into 3 main groups (a) 2,3,7,8-T_4CDD and about 12 other extremely toxic congeners considered equitoxic. (b) Mono-, di-, tri-, and octa-chlorinated congeners combined into low toxicity group and prorated or ignored. (c) All other congeners considered to be of intermediate toxicity and prorated according to the geometric mean of the most toxic forms.	- increased realism - attainable - MOE already ignores mono-, di- and tri-chlorinated congeners	- requires isomer-specific analysis - requires full range of PCDD and PCDF standards - increased computational problems - decreased safety factor
(5) Divide congeners into isomer groups and prorate each isomer group according to most toxic isomer (2,3,7,8-T_4CDD treated seperately and next most toxic T_4CDD used for that isomer group)	- high level of realism - attainable - analytically measured masses of isomer groups are available	- requires isomer-specific analysis for 2,3,7,8-T_4CDD - requires full range of representative PCDD and PCDF standards - increased computational problems - decreased safety factor - difficulty prorating toxicity
e.g. 2,3,7,8-T_4CDD + $\left[\frac{M_1CDD}{X} + \frac{D_2CDD}{Y} + \ldots + \frac{O_8CDD}{Z} + \frac{M_1CDF}{X_2} + \frac{D_2CDF}{Y_2} + \ldots + \frac{O_8CDF}{Z_2}\right]$ where X, Y, Z, X_2, Y_2, etc. are prorating factors.		
(6) Use biological equivalents	- most realistic	- untested - validation problems

biological tissues. Where toxicity relationships can be derived from the limited number of carcinogenicity and teratology studies these data are used.

Assumptions which lead to the different proposed toxicity prorating factors for each PCDD or PCDF isomer group are as follows:

Dibenzo-p-dioxin (DD) and Dibenzofuran (DF)

- parent molecules
- no LD_{50} data
- inactive in *in vitro* AHH induction assay
- no carcinogenic response in lifetime studies of rats and mice fed 10,000 ppm DD in their diet
- not considered toxic.

M_1CDDS

- 2 isomers
- no LD_{50} data
- no *in vitro* AHH induction data
- no fetal anomalies in litters of Wistar rats fed 2,000 ug 2-M_1CDD/kg/day during gestation days 6 to 15. Comparable NOEL for 2,3,7,8-T_4CDD is 0.03 ug/kg/ day suggesting that M_1CDDs are 67,000-fold less teratogenic than 2,3,7,8-T_4CDD.
- toxicity assumed less than or comparable to O_8CDD (see below).

D_2CDDs

- 10 isomers
- LD_{50} (guinea pig) for 2,8-D_2CDD is 150,000-fold less toxic than 2,3,7,8-T_4CDD
- 2,8-D_2CDD not active in *in vitro* AHH induction test (25,400-fold less active than 2,3,7,8-T_4CDD)
- slight myocardial oedema in fetuses of Wistar rats fed 500 ug 2,7-D_2CDD/kg/day on gestation

days 6 to 15. Comparable LEL for 2,3,7,8-T_4CDD in rats is 0.125 ug/kg/day suggesting that 2,7-D_2CDD is 4,000-fold less teratogenic than 2,3,7,8-T_4CDD.

- suggestive carcinogenic effect in male B6C3F1 mice fed 10,000 ppm 2,7-D_2CDD in diet (estimated at 1,000 ug 2,7-D_2CDD/kg/day). Comparable LEL for 2,3,7,8-$_4$CDD in male B6C3F1 mice is 0.07 ug/kg/day suggesting that 2,7-D_2CDD is about 14,000-fold less carcinogenic than 2,3,7,8-T_4CDD.
- assume D_2CDD isomers have a toxicity of 0.1% (i.e. 0.001) compared with 2,3,7,8-T_4CDD.

T_3CDDs

- 14 isomers, 2 potentially toxic
- LD_{50} 2,3,7-T_3CDD is greater than 11-fold (mouse) to 15,000-fold (guinea pig) less toxic than 2,3,7,8-T_4CDD
- *in vitro* AHH induction activity (2,3,7-T_3CDD) is 157-fold to 2,750-fold less active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume T_3CDD isomers have a toxicity of 1.0% (i.e. 0.01) compared with 2,3,7,8-T_4CDD

T_4CDDs

- 22 isomers, 5 (including 2,3,7,8-T_4CDD) potentially toxic
- toxicity of 2,3,7,8-T_4CDD well established
- other T_4CDD isomers treated separately from 2,3,7,8-T_4CDD
- no LD_{50} data for other T_4CDD isomers
- *in vitro* AHH induction, most active isomer (1,3,7,8-T_4CDD) is 185-fold to 240-fold less active than 2,3,7,8-T_4CDD

- no fetal or postnatal effects in progeny of rats treated with 800 ug 1,2,3,4-T_4CDD/kg/day during gestation day 6 to 15. Comparable NOEL for 2,3,7,8-T_4CDD is 0.03 ug/kg/ day suggesting 1,2,3,4-T_4CDD is 27,000-fold less toxic than 2,3,7,8-T_4CDD.
- no carcinogenicity data for other T_4CDD isomers
- treat 2,3,7,8-T_4CDD separately (100% toxicity)
- assume other T_4CDD isomers have a toxicity of 1% (i.e. 0.01) compared with 2,3,7,8-T_4CDD

P_5CDDs

- 14 isomers, 7 potentially toxic
- LD_{50} (1,2,3,7,8-P_5CDD) 1.2-fold (mouse) to 1.5-fold (guinea pig) less toxic than 2,3,7,8-T_4CDD
- in vitro AHH induction (1,2,3,7,8-P_5CDD) is 17-fold less active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume P_5CDD isomers have a toxicity of 10% (i.e. 0.1) compared with 2,3,7,8-T_4CDD

H_6CDDs

- 10 isomers, 7 potentially toxic
- LD_{50} (1,2,3,4,7,8-H_6CDD) 3-fold (mouse) to 36-fold (guinea pig) less toxic than 2,3,7,8-T_4CDD
- in vitro AHH induction (1,2,3,4,7,8-H_6CDD) 25-fold to 80-fold less active than 2,3,7,8-T_4CDD
- the NOEL for embryotoxicity and teratogenicity for mixtures of H_6CDDs in rats is 0.1 ug/kg/day during fetal development, the comparable NOEL for 2,3,7,8-T_4CDD is 0.03 ug/kg/day suggesting that mixtures of HCDDs are 30-fold less teratogenic than 2,3,7,8-T_4CDD
- the NOEL for carcinogenicity for a mixture of 1,2,3,6,7,8-H_6CDD and 1,2,3,7,8,9-H_6CDD in rats is 1.25 ug/kg/day, the comparable NOEL for

2,3,7,8-T_4CDD is 0.001 ug/kg/day suggesting that mixtures of H_6CDDs are 1,250-fold less carcinogenic than 2,3,7,8-T_4CDD

- assume H_6CDD isomers have a toxicity 10% (i.e. 0.1) compared with 2,3,7,8-T_4CDD

H_7CDDs

- 2 isomers, both potentially toxic
- LD_{50} (1,2,3,4,6,7,8-H_7CDD) guinea pig > 300-fold less toxic than 2,3,7,8-TCDD
- *in vitro* AHH induction (1,2,3,4,6,7,8-H_6CDD) 500-fold less active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume H_7CDD isomers have a toxicity of 1.0% (i.e.) 0.01 compared with 2,3,7,8-T_4CDD

O_8CDD

- 1 isomer
- LD_{50} (rat and mouse) >500,000-fold less toxic than 2,3,7,8-T_4CDD
- *in vitro* AHH induction >46,000-fold less active than 2,3,7,8-T_4CDD
- no fetal anomalies in rats fed 100,000 ug O_8CDD/kg/day during fetal development. Comparable NOEL for 2,3,7,8-T_4CDD is 0.03 ug/kg/day suggesting that O_8CDD is at least 3,000,000-fold less teratogenic than 2,3,7,8-T_4CDD.
- no carcinogenicity data
- assume O_8CDD has a toxicity of 0.01% (i.e. 0.0001) compared with 2,3,7,8-T_4CDD

M_1CDFs

- 4 isomers
- no LD_{50} data
- *in vitro* AHH induction (4-M_1CDF) about 1,000,000-fold less active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data

- assume M_1CDF isomers have a toxicity less than or equivalent to O_8CDD (see above)

D_2CDFs

- 16 isomers
- no LD_{50} data
- *in vitro* AHH induction (2,3-D_2CDF) 260,000-fold less active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume D_2CDF isomers have a toxicity less than or equal to O_8CDD (see above)

T_3CDFs

- 28 isomers, 4 potentially toxic
- no LD_{50} data
- *in vitro* AHH induction (2,3,8-T_3CDF) 1,600-fold less active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume by analogy with T_3CDDs that T_3CDFs have a toxicity 1% (i.e. 0.01) compared with 2,3,7,8-T_4CDD

T_4CDFs

- 38 isomers, 8 potentially toxic
- LD_{50} data (2,3,7,8-T_4CDF) one third as toxic as 2,3,7,8-T_4CDD
- *in vitro* AHH Induction (2,3,7,8-T_4CDF) 8-fold less active as 2,3,7,8-T_4CDD
- *in vivo* AHH induction (2,3,7,8-T_4CDF) about 60% as active as 2,3,7,8-T_4CDD
- NOEL for teratogenicity in C57BL/6 mice is 10 ug/kg/day during fetal development, comparable NOEL for 2,3,7,8-T_4CDD is 0.1 ug/kg/day suggesting that 2,3,7,8-T_4CDF is 100-fold less teratogenic than 2,3,7,8-T_4CDD
- no carcinogenicity data

- assume T_4CDF isomers have a toxicity of 50% (i.e. 0.5) compared with 2,3,7,8-T_4CDD

P_5CDFs

- 28 isomers, 14 potentially toxic
- LD_{50} (2,3,4,7,8-P_5CDF) one third as toxic as 2,3,7,8-T_4CDD
- *in vitro* AHH induction (2,3,4,7,8-P_5CDF) 45-fold less toxic than 2,3,7,8-T_4CDD
- *in vivo* AHH induction (2,3,4,7,8-P_5CDF) about 30% as active as 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume that P_5CDF isomers have a toxicity of 50% (i.e. 0.5) compared with 2,3,7,8-T_4CDD

H_6CDFs

- 16 isomers, 12 potentially toxic
- LD_{50} (2,3,4,6,7,8-H_6CDF) 60-fold less toxic than 2,3,7,8-T_4CDD
- *in vitro* AHH Induction (1,2,3,4,7,8-H_6CDF) 65-fold lass active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume that H_6CDF isomers have a toxicity of 10% (i.e. 0.1) compared with 2,3,7,8-T_4CDD

H_7CDFs

- 4 isomers, 2 potentially toxic
- no LD_{50} data
- *in vitro* AHH induction (1,2,3,4,6,8,9-H_7CDF) 120,000-fold less active than 2,3,7,8-T_4CDD
- no teratology or carcinogenicity data
- assume by analogy with H_7CDD isomers that H_7CDF isomers have a toxicity of 1% (i.e. 0.01) compared with 2,3,7,8-T_4CDD

O_8CDF

- one isomer
- no toxicity or enzyme induction data available
- assume by analogy with O_8CDD that O_8CDF has a toxicity of 0.01% (i.e. 0.0001) compared with 2,3,7,8-T_4CDD

In summary, the following prorating factors relative to 2,3,7,8-T_4CDD are proposed:

TABLE 3.6.7C

ESTIMATED RELATIVE TOXICITY OF PCDD AND PCDF ISOMERS TO 2,3,7,8-T_4CDD

Isomer Groups	Toxicity factor relative to 2,3,7,8-T_4CDD
DD	non toxic
M_1CDD	0.0001
D_2CDD	0.001
T_3CDD	0.01
T_4CDD*	0.01
P_5CDD	0.1
H_6CDD	0.1
H_7CDD	0.01
O_8CDD	0.0001
DF	non toxic
M_1CDF	0.0001
D_2CDF	0.0001
T_3CDF	0.01
T_4CDF	0.5
P_5CDF	0.5
H_6CDF	0.1
H_7CDF	0.01
O_8CDF	0.0001

* excluding 2,3,7,8-T_4CDD

Summary

Development of the relative toxicity factors of the various PCDD and PCDF isomers compared to 2,3,7,8-T_4CDD is based on a review of current research data.

3.7 ESTIMATION OF MAXIMUM ALLOWABLE DAILY INTAKE

3.7.1 INTRODUCTION (SUMMARY OF HAZARD IDENTIFICATION AND SELECTION OF CRITICAL TOXICOLOGICAL CRITERIA)

In the preceding sections of this chapter hazards to human or environmental health have been examined and the dose-response relationship (where available) has been assessed.

The toxicological information is based mainly on studies of effects of 2,3,7,8-T_4CDD. This is the major toxic contaminant of 2,4,5- T_3CP derived products or waste.

However, 2,3,7,8-T_4CDD is only a minor trace component of PCDD/PCDF mixtures resulting from refuse incineration and is not found in chlorophenol wood preservatives, 2,4-D herbicides or PCBs. In Ontario these products or processes are the major sources of PCDDs and PCDFs not 2,4,5-T_3CP derived products or wastes. Thus toxicological data on the congeners which occur in the greatest concentrations and distribution in the Ontario environment are absent.

The proposed standard therefore will be based on 2,3,7,8-T_4CDD but must address the toxic contributions of other PCDDs and PCDFs found in the Ontario environment.

Summary of 2,3,7,8-T_4CDD Toxicology

1. Human Health Effects

Acute exposure to 2,3,7,8-T_4CDD results in symptoms (e.g., chloracne) which slowly decrease with time. Outside of occupational or accidental exposure situations, PCDDs/PCDFs are not present in the environment at levels capable of causing acute effects in man. The evidence for a chronic human health effect relating to cancer, birth defects, coronary heart disease, or impairment of the immune system is inconclusive and requires further investigation.

Therefore, it is recommended that the development of environmental standards for PCDD and PCDFs should be based on animal toxicological studies.

2. Animal Studies

a) In acute toxicity studies, 2,3,7,8-T_4CDD is classed as extremely toxic ($LD_{50} < 500$ ug/kg) even though wide ranges of sensitivities were found in species tested (LD_{50} 0.6 to 5,000 ug/kg). Rats and monkeys are only 20-fold less sensitive than the most sensitive species, guinea pig.

Multiple toxic effects are found in various organ systems ("wasting" syndrome).
In this syndrome, death is delayed. Some overlap of symptoms between acute and chronic studies is found.

This extreme acute toxicity should be considered when assessing carcinogenic and teratogenic potencies.

b) Data are available from long-term/lifetime animal studies of carcinogenicity.

Increased incidence of certain spontaneous tumours is observed in rats and mice. Significant tumour induction was generally demonstrated in both species over the range 0.007 to 0.3 ug 2,3,7,8-T_4CDD/kg/day. Onset of tumour appearance was delayed usually to the latter half of the lifetime study.

Those studies involving oral administration (2,3,7,8-T_4CDD is most toxic using this route of exposure) have reliable dose-response data and indicate clear NOEL (0.001 to 0.0014 ug/kg/day) for tumour incidence.

c) Data from teratological (reproductive effects) studies in rats and mice indicate that 2,3,7,8-T_4CDD causes embryolethality and/or fetotoxicity. A 3-generation reproduction study of rats was associated with decreased fertility and neonatal survival in the first and second generations.

NOELs for these teratogenic reproductive effects over the range 0.001 to 0.03 ug 2,3,7,8-T_4CDD/kg have been reported.

3. Genetic Toxicology

2,3,7,8-T_4CDD is non-mutagenic in in vitro microbial mutagenicity tests, non-clastogenic in *in vivo* mammalian tests and does not form covalent bonds with DNA. 2,3,7,8-T_4CDD is an unconfirmed mutagen in one *in vivo* mammalian mutagenicity test. However, it is concluded here that it is not a classical mutagen that reacts directly with DNA.

4. Environmental Toxicology

Although environmental contamination and subsequent adverse effects on animals have been documented following localized accidental release of PCDDs or PCDFs, no long-term toxic ecosystem or species effects have been shown. Potentially sensitive species appear to be embryonic and larval forms of fish and fish-eating birds. Some fish appear to be sensitive to very low concentrations (ng/L) of 2,3,7,8-T_4CDD.

5. Toxicology of other PCDDs and PCDFs

Acute toxicities of other PCDDs and PCDFs range up to 10^5-fold less than 2,3,7,8-T_4CDD.

Structure-activity relationships indicate that PCDDs and PCDFs are most toxic when there are four laterally substituted halogen atoms and at least one vicinal hydrogen atom present on the structure. This structure correlates well with the ability to bind to the cytosolic receptor, to induce AHH enzymes, and the potential for acute toxicity.

PCDDs and PCDFs elicit similar toxic effects, with the T_4CDD, T_4CDF, P_5CDD and P_5CDF congeners chlorinated in the 2,3,7 and 8 positions being the most toxic. Mon-, di- and octa-chlorinated congeners are the least toxic.

These structure-activity relationships in conjunction with other toxicity data can be used to prorate the toxicity of PCDDs and PCDFs in terms of "2,3,7,8-T_4CDD equivalents".

6. Supporting Evidence

PCDDs and PCDFs are environmentally persistent, usually bound to organic or inorganic matrices. Evidence indicates that bound PCDDs and PCDFs vary in bioavailability and the more toxic forms are slowly eliminated or degraded following uptake. PCDDs and PCDFs appear to be more active in their "native" form than as metabolites.

Ranking of PCDDs and PCDFs

The information summarized above can be used to rank PCDDs/PCDFs as toxic agents, principally as carcinogenic hazards based on animal studies only, so that informed decisions may be made to protect human health. Several ranking schemes have been proposed, the most recent being that of Theiss (1983). These ranking schemes are based on either the dose required to cause a carcinogenic response (ACGIH; 1982), the mechanism of carcinogenesis involved (Weisburger and Williams, 1980; ECETOC, 1982; Pitot, 1983); or the magnitude of the carcinogenic response following exposure (Theiss, 1983; Squire, 1981; IARC, 1983).

Cancer is a large group of related diseases in which cells fail to observe the normal signals which regulate cell proliferation. At least some cancers arise from direct genetic changes at the DNA level, e.g. mutations, which occur in somatic cells. Studies of transforming genes and oncogenes indicate that chromosomal rearrangements may be as important as classical mutagenic events in the initiation of cancer. Many different mechanisms may be important in the initiation and development of cancer.

A wide variety of indirect non-mutational(or epigenetic) mechanisms may also act to facilitate development of initiated cells.

Several models of carcinogenesis have been proposed following studies of cancer in humans or animals exposed to radiation or chemicals. Consequently, a substance can be called a complete carcinogen, an initiator, a promoter, or a co-carcinogen. The whole animal bioassay is the closest surrogate for measures of human carcinogenicity. However, most whole animal bioassays do not distinguish between so-called initiating or promoting agents.

Based upon such differences in the biological activities of carcinogens, a classification of carcinogens as either genotoxic or epigenetic (nongenotoxic) has been suggested (Weisburger and Williams, 1981). This has also been suggested as an approach to risk assessment and it has been proposed that thresholds for nongenotoxic carcinogens probably exist because their effects are dose-dependent and reversible. However, others have argued against assuming no-effect levels for any type of animal carcinogen, since the actual carcinogenic mechanisms are not known for any substance, and because the absence of low-level risks cannot be proven by any existing methods. Furthermore, a recent report by The International Agency for Research on Cancer (IARC, 1983b) indicates that the present evidence does not warrant classification of carcinogens based upon mechanisms of action such as genotoxicity.

There appears to be some agreement that it is not prudent at present to base risk assessment determinations solely upon genotoxicity anymore than any other single factor. Too many uncertainties remain in our understanding of neoplastic transformation and development. In any event, chemicals apparently do not have to be genotoxic to act as so-called "complete carcinogens." Although it may be conceded that the absence of low-level risk cannot be directly proved, it must be acknowledged that this holds for all toxicological effects. Cancer is not unique in this regard. What is unique is our hypothetical assumption that carcinogenic transformation results from irreversible, low-level, one-hit type events. Even if this were the case, such phenomena would be limited to chemicals or their metabolites which can directly alter DNA.

For decision-making purposes, chemicals can be ranked on the basis of chronic, irreversible effects, such as carcinogenicity, mutagenicity or effects on reproduction, following exposure: 1) in terms of human data; 2) based on acute or chronic animal tests; and 3) short-term tests for genotoxicity. (Section 3.1)

In the case of PCDDs and PCDFs, carcinogenicity is assumed to be the most sensitive toxic end point and the most toxic and most studied PCDD or PCDF congener is 2,3,7,8-T_4CDD (Section 3.7.2).

1. Human data are being developed from ongoing large-scale epidemiological studies. Existing human epidemiological data and data from acute human exposure do not allow the determination of a safe level of exposure. Therefore, data from the studies with mammalian species cited above must be used to estimate the chronic health risk of PCDD or PCDF exposure to humans.

2. 2,3,7,8-T_4CDD has been associated with an increased incidence of certain spontaneous tumours in laboratory rats and mice when given orally in the diet, by gavage or cutaneously over a two year period. In the absence of any human data indicating greater sensitivity to PCDDs or PCDFs than the rodent species used in lifetime carcinogenicity studies and, considering that the acute toxicities of 2,3,7,8-T_4CDD to these rodents overlap the LD_{50} for the rhesus monkey, it is assumed that rodent data are prudent bases for estimating the chronic health risks of PCDD or PCDF exposure to humans.

   The NOEL identified in rodent carcinogenicity bioassay studies indicate where the threshold level for tumour production by 2,3,7,8-T_4CDD exists.

While 2,3,7,8-T_4CDD has been rated as the most potent carcinogen using absolute quantities as a criterion this must be viewed in the context that it is also the most acutely toxic synthetic chemical known to man and its carcinogenic properties are expressed at concentrations only 2 to 3 orders of magnitude below the LD_{50} range and NOELS are found only 4 to 5 orders of magnitude below the LD_{50} range.

3. Mutagenicity studies, judged on a battery of short-term tests, indicate that 2,3,7,8-T_4CDD is not a mutagen in the classical sense. The lack of evidence to suggest that 2,3,7,8-T_4CDD or its metabolites can directly alter DNA physically or chemically also supports this conclusion.

From the data in the above section, it can be concluded that PCDDs and PCDFs, especially 2,3,7,8-T_4CDD, can produce tumours in rodents by an indirect mechanism. A threshold dose exists as indicated by NOELS from long term animal studies.

3.7.2 HAZARD ESTIMATION

Hazard estimation is the phase of the hazard assessment process when the quantitative aspects of the toxicological data base are assessed so that an adequate margin of safety can be incorporated into the proposed safe level of intake. Quantitative data for dose-response relationships and the lowest (LEL) or no effect (NOEL) levels must be considered in terms of the most sensitive response (endpoint) and most sensitive species.
(Table 3.7.2A)

TABLE 3.7.2A
LOWER RESPONSE LIMITS FOR 2,3,7,8-T_4CDD IN RODENTS

RESPONSE (ENDPOINT)	2,3,7,8-T_4CDD CONCENTRATION	REFERENCE
Biological activity (AHH induction) - rat	0.002 ug/kg (LEL)	Kitchin and Wood, 1978
Tissue toxicity (chloracne) - rabbit	0.01 - 0.1 ug/kg (ED_{50})	Schwetz *et al.*, 1973
(hepatocellular changes) - rat	0.01 ug/kg/day (LEL)	Kociba *et al.*, 1978
Reproductive effects (teratogenicity) - mouse	0.03 ug/kg/day (LEL)	Murray *et al.*, 1979
Tumour production - rat	0.001 ug/kg/day (NOEL)	Kociba *et al.*, 1978

While rats and mice are at least 10-fold less sensitive to 2,3,7,8-T_4CDD than guinea pigs (the most sensitive mammalian species) on an acute lethality basis, their sensitivity to 2,3,7,8-T_4CDD overlaps with non-human primates (Table 3.2.2A).

Man has been exposed to phenoxy herbicides, chlorinated phenols, polychlorinated biphenyls and other chemicals containing PCDDs/PCDFs over a number of years. Apart from acute exposure episodes involving highly contaminated materials (Seveso, Times Beach, Binghamton State Office Building, Yusho incident, and numerous industrial accidents), no adverse human health effects have been observed. These observations suggest that man is no more sensitive to 2,3,7,8-T_4CDD than rats or mice.

Therefore, it is appropriate to use these data in human health risk estimation.

In the absence of reliable human epidemiological data demonstrating a cause and effect relationship following well-characterized exposure to a chemical, two main approaches are recommended to estimate non-toxic or safe levels of exposure: safety (uncertainty) factors and quantitative risk analysis.

> Safety (uncertainty) factors can be applied to chemicals exhibiting NOEL and can be reasonably assumed to have exposure thresholds below which the chance of harm is remote.
>
> Quantitative risk analysis involves the use of mathematical models to extrapolate from dose-response data to virtually safe dose (VSD) levels.

Both involve analysis of dose-response data from long-term animal bioassay results.

Safety factors usually incorporate several internal factors to account for: 1) intraspecies variability, i.e., more sensitive members of the species not measured in the sample of test animals; and 2) interspecies adjustment-extrapolation from animal exposure data to humans. This latter factor can be based on the body weight, surface area, the surface area to body weight ratio, the percentage of intake as a fraction of body weight, or as a percentage of diet or water.

Criticisms of the use of the safety factor approach for setting recommended daily intakes for PCDDs/PCDFs are:

a) definition of NOEL - the precision of this measurement is proportional to the number of animals tested. However, an intraspecific variability factor of 10 would seem adequate to account for the uncertainty involved here;

b) slope of dose-response curve not accounted for - in the case of the available lifetime rodent bioassay data for 2,3,7,8-T_4CDD, only three non-linear data points are available for extrapolation;

c) traditionally, safety factors have been based on acute rather than chronic toxicity data and generally are not used with direct acting carcinogens - in the case of PCDDs and PCDFs, especially the most toxic form, 2,3,7,8-T_4CDD, the evidence indicates that its association with tumour production is indirect and that a threshold, as indicated by NOEL data, exists;

d) problem of accumulation - safety factors may be inappropriate for substances that accumulate in the tissues, e.g. heavy metals or certain lipophilic organic compounds. Comparison of the integrated lifetime dose (ug/kg) based on the lifetime average dose (ug/kg/day) in rodent studies indicates that only a fraction of the 2,3,7,8-T_4CDD administered over the lifetime of the animal is retained at the NOEL level of exposure.

Quantitative risk analysis (low dose extrapolation models) do have several advantages in that they take the slope of the dose-response curve into account, they can be used for non-threshold substances or processes, and they can mathematically relate level of exposure to degree of risk (or probability of a toxic response). Problems associated with the use of mathematical models include:

a) the multiplicity of models and their ability to produce very different estimates of risk from the same biological data;

b) the fact that current models only assume the direct action of the chemical or process; assumptions about indirect modes of action are presently not incorporated into these models; and

c) current models extrapolate probabilities from measurements in the 10^{-1} to 10^{-2} range down to 10^{-5}, 10^{-6} or 10^{-8}, i.e., well beyond the realm of biological certainty.

3.7.3 LOW DOSE EXTRAPOLATION (MATHEMATICAL MODELS)

The use of quantitative risk-analysis models is well reviewed. Only two chronic feeding studies from the available published literature can be used for risk-analysis calculations. Risk analysis of the dose-response data from these studies using models which assume a direct, genotoxic mode of action have been performed by National Research Council of Canada, the U.S. Environmental Protection Agency, (EPA) and the Centre for Disease

Control in Atlanta (CDC). A summary of the results obtained for various levels of risk are tabulated in Table 3.7.3A. Since similar models have been applied to the same data base, the range of VSD's obtained is similar. Some overlap can be seen between the maximum allowable daily intake derived using the safety factor approach (Section 3.7.4) and some of the models used at the 10^{-4} and 10^{-6} calculated levels of risk. The non-linear and sex-specific nature of the rodent bioassay data used and the presence of dose-related primary liver damage at treatment levels causing hepatocellular neoplastic changes (Kociba et al., 1978, 1979; NTP, 1980) suggest that these risk estimates for cancer incidence may be confounded by direct tissue damage. Lack of knowledge of the mode of action of 2,3,7,8-T_4CDD also precludes selection of a specific risk analysis model. **Use of these risk-analysis models, in this instance, should therefore be more to indicate the potential range of safe doses rather than to form the basis of a standard.**

TABLE 3.7.3A

ESTIMATES AND APPROXIMATE LOWER CONFIDENCE LIMITS FOR THE VIRTUALLY SAFE DOSE (VSD) OF 2,3,7,8-T_4CDD AT VARIOUS RISK LEVELS

VSD (LOWER DOSE AT 95% CONFIDENCE LIMIT)*
(pg/kg body weight/day)

RISK LEVEL**	10^{-4}	10^{-6}	10^{-8}
NRCC			
Multistage model	8.2(6.5)	0.09(0.07)	0.09(0.00007)
Multistage model (omitting top dose)	176(2.2)	17.6(0.02)	17.6(0.00002)
Weibull model	107(0.00005)	0.12(4×10^{-8})	1.31×10^{-7}(3×10^{-12})
Linear model (from 0.01 ug/kg/day)	3.9(2.6)	0.04(0.03)	0.4×10^{-4}(0.3×10^{-4})
Linear model (from 0.001 ug/kg/day)	N.D.(3.1)	N.D.(0.03)	N.D.(0.3×10^{-4})
CDC			
Linear model	7.7(5.7)	0.08(0.06)	N.D.
Transformed multistage model	3.8(2.9)	0.04(0.03)	N.D.

* based on hepatocellular neoplasms and carcinoma in female rats (data of Kociba et al., 1978).

** Estimated level of additional cancer incidence in the population following lifetime exposure to the indicated VSD.

3.7.4 ESTIMATION OF MAXIMUM ALLOWABLE DAILY INTAKE

Use of either the NOEL/Safety Factor or the Low Dose Extrapolation approach to risk estimation is arbitrary and very much dependent on the quality of the biological data available. It is the recommendation of the authors based on their scientific judgement of the evidence reviewed in this report that the safety factor approach may be used when setting an adequate margin of safety to protect the citizens of Ontario.

Depending on the level of toxicological information available, the following safety factors have been used:

- a factor of 10 when chronic human exposure data are available and are supported by chronic oral toxicity data in animal species;
- a factor of 100 when good chronic oral toxicity data are available for some animal species but not in humans, and;
- a factor of 1000 with limited chronic animal toxicity data (National Academy of Sciences, 1977).

It should be cautioned that these safety factors only represent judgements regarding allowable levels of human exposure and are not guarantees of absolute safety.

PCDDs and PCDFs have similar toxicities, with the 2,3,7,8-substituted congeners in both groups being the most toxic. The 2,3,7,8-T_4CDD congener is the most toxic and most studied of the group and is used to derive a standard for all congeners in terms of their equivalent toxicity to it.

From preceeding sections, it is concluded that 2,3,7,8- T_4CDD produces tumours in rodents by an indirect mechanism. As such, it is concluded that 2,3,7,8- T_4CDD possesses a threshold level of dosage below which an increased rate of tumour production due to exposure to this compound would be unlikely.

The no-observed effect level (NOEL) identified in animal studies is used as an indication of where the threshold lies. A maximum allowable daily intake can be based on this level by applying a safety factor. In the case where long-term animal studies are available, it is generally agreed that a safety factor of 100 is appropriate (Nat. Acad. Sci., 1977).

This 100-fold safety factor is a practical means to handle the uncertainties in extrapolating from animals to humans. It includes a factor of 10 to extrapolate from animals to humans assuming that animals are less sensitive than humans and another factor of 10 to account for differential sensitivities within the human population.

This factor incorporates a number of considerations to account for uncertainty in extrapolating from animal data to humans, particularly an allowance in case humans are more sensitive than the animal species tested.

Since acute toxicity and long-term animal studies are available, and since the short-term mutagenicity studies, and the human epidemiology studies are generally negative, a safety factor of 100 is recommended.

The NOEL of 0.001 ug/kg/day for 2,3,7,8-T_4CDD, determined in the three-generation reproductive study of Murray et al. (1979) and the two-year oncology study of Kociba et al. (1978) both using rats, is recommended as a prudent basis for developing a maximum allowable daily intake for human PCDD and PCDF intake.

Thus using the NOEL of 0.001 ug/kg/day and a safety factor of 100 yields a maximum allowable daily intake of 1 x 10^{-5} ug/kg/day for humans. The dose from all sources for a 60 kg person would be 60 x 10^{-5} ug/ day.

The recommended maximum allowable daily intake for total PCDDs and PCDFs is the equivalent of 10 pg 2,3,7,8-T_4CDD/kg body weight/day not to be exceeded on average over a year.

4.0 SOURCES/INPUTS, DISTRIBUTION AND FATE OF PCDDs AND PCDFs IN THE ENVIRONMENT

4.1 INTRODUCTION

The exposure evaluation process of the risk analysis procedure (Chapter 5) requires an assessment of the sources, inputs and resulting concentrations of PCDDs and PCDFs in environmental media. Information gathered from the source assessment process can be used in three ways;

1. To identify, quantify and rank processes that input PCDDs and PCDFs to the Ontario environment, and,
2. To identify, quantify and rank the resulting concentrations of PCDDs and PCDFs in environmental media following distribution.
3. To determine natural or existing background levels of PCDDs and PCDFs in environmental media.

This chapter will review sources and processes previously identified in the literature such as;

1) Combustion sources (section 4.2.1),
2) Chemical manufacturing and use (section 4.2.2),
3) Waste Disposal Sites, (section 4.2.3) and,
4) Transboundary sources (section 4.2.4).

Estimated quantities of PCDDs and PCDFs from each of these sources or processes are summarized in Section 4.2.5

The environmental transport, distribution and fate of PCDDs and PCDFs released into the environment are reviewed in Section 4.3. Ontario data on PCDD and/or PCDF residues in environmental media are limited to:

1) Calculated levels of PCDDs and PCDFs in ambient air near incinerators (Section 4.2.1.7)
2) Levels of PCDDs and PCDFs in soils near industrial sites (Section 4.3.2.1)
3) Levels of PCDDs and PCDFs in fish from the Great Lakes (Section 4.3.4.4)
4) Levels of PCDDs and PCDFs in Ontario drinking water supplies (Section 4.2.4.4), and
5) Levels of PCDDs and PCDFs in groundwater near industrial sites (Section 4.2.3.1)

Supplemental information on the environmental transport, distribution and fate of PCDDs and PCDFs is derived from experimental data or data obtained from PCDD- and/or PCDF- contaminated sites outside of Ontario.

4.2 SOURCES/INPUTS

4.2.1 COMBUSTION SOURCES

4.2.1.1 Municipal Waste Incineration

Since the first published report of PCDD and PCDF emissions from municipal incinerators (Olie *et al.*, 1977) a large number of studies have been carried out to examine this phenomenon. In the Canadian NRCC document (1981a) many of the studies which had been published as of March 1981 were cited and included work by Buser *et al.* (1978a, 1978b); Eiceman *et al.* (1979); Karasek (1980); Bumb *et al.* (1980); Cavallaro *et al.* (1980); Lamparski and Nestrick (1980); Tiernan (1980) and Lustenhouwer *et al.* (1980). Some of the main factors concerning the emission of PCDD and PCDF from these sources which were identified and described in the NRCC report are presented below.

(1) all of the chlorinated isomer groups (T_4CDD-O_8CDD and T_4CDF-O_8CDF) have been detected in fly ash and in flue gas particulate matter in highly variable quantities.

(2) the emitted particulate matter has a higher PCDD concentration than that precipitated by the fly ash collectors (estimated at 10x).

(3) the reported fly ash levels for Ontario incinerators appear lower than those from similar sources in the Netherlands; the congener pattern also is different.

(4) on an isomer-specific basis an evaluation of fly ash from U.S. and European incinerators revealed that at least 14 of 22 possible T_4CDD isomers were present with very similar ratios

being detected between the various isomers relative to total T_4CDDs for both sources.

(5) the main T_4CDD isomers in the U.S. and European fly ash included 1,3,7,8-, 1,3,7,9-, 1,3,6,8-, 1,2,3,7- and 1,2,3,8-T_4CDD.

(6) At least 7 of the 10 possible H_6CDD isomers also were detected in varying degrees in the fly ash.

As many additional studies have been published in the three years since the release of the NRCC document (1981a), it is now possible to assess more specifically the formation and release of PCDDs and PCDFs from the incineration of raw and treated municipal waste and to re-evaluate some of the emission characteristics identified in the earlier NRCC document. This review and analysis of the global data base will provide a comparative basis for the subsequent analysis of the contribution from Ontario sources (Sec.4.2.1.7).

Formation Processes and Operational Conditions

The recent literature (Lustenhouwer *et al.*, 1980, Liberti *et al.*, 1983, Choudhry *et al.*, 1982) generally considers three possibilities to account for the presence of PCDDs and PCDFs in flue gas and fly ash emissions from municipal and industrial incinerators.

(1) PCDDs/PCDFs are already present in the industrial and domestic waste.

(2) PCDDs/PCDFs are produced directly from chlorinated precursors which are already present in the refuse.

(3) PCDDs/PCDFs are formed in the incinerators through *de novo* synthesis from chemically unrelated (non-chlorinated) organic matter.

In their theoretical approach to resolve the issue of PCDD/PCDF formation Choudhry et al. (1982) examined the latter two mechanisms, and concluded that there is no doubt that de novo synthesis is involved in the formation of PCDDs and PCDFs in incinerators and other combustion sources. They also theorized that chlorobenzenes, chlorophenols and PCBs are key intermediate compounds in the de novo synthesis of PCDDs or PCDFs from the reaction of organic matter in the presence of oxygen and chlorine. They cite the data of Lustenhouwer et al. (1980) and Liberti et al. (1978) to indicate that although PCDDs and PCDFs could enter municipal refuse via herbicidal formulations, treated wood, or PCB-containing products the analytical data on raw refuse analyzed to date, do not support this route of entry as a significant source. The combustion of major PCDD and PCDF chlorinated precursors which could already be present in the refuse also was believed to be of minor importance in relation to de novo synthesis.

These negative raw refuse analyses results are not supported by the recent Ontario data (Ozvacic et al., 1984a) in which average concentrations of 19.8ng/g PCDDs (T_4-O_8), 2.3ng/g PCDFs (T_4-O_8), 12.6ng/g chlorobenzenes, 79.8ng/g PCBs, and 521.3ng/g chlorophenols were confirmed in raw municipal refuse,(dry weight basis).

While other authors have taken a more experimental approach to resolving the formation process by incorporating or removing different precursor materials into municipal refuse before or during the incineration process, the results which have been published tend to support the theoretical work of Choudhry et al. (1982). Included in these studies are those of Liberti and Brocco (1982) and Liberti *et. al.* (1983) who conceptualize the *de novo* synthesis theory as:

Precursor A	+	Precursor B	=>	PCDD/PCDF
phenols polyphenols or compounds with phenol structures		compounds containing chlorine or HCl donors		

Olie *et al.* (1983) in micro-scale studies involving the pyrolysis of lignin and chlorine donors as well as the incineration of municipal refuse with additional hexachlorobenzene, add further support to the requirement for a lignin or lignin-like structure plus some form of chlorine donor in the formation of PCDDs/PCDFs.

Although the stack gas analysis work of Tiernan *et al.* (1983b) does not entirely support the *de novo* theory, the authors do conclude that there probably are several formation mechanisms involving both chlorobenzenes and chlorophenols. This is based on the absence of a straight-forward chemical concentration relationship between the various chlorophenol and chlorobenzene congeners in the stack gases and the concentration of the corresponding PCDD/PCDF chlorinated congeners. They also point out that the relatively low levels of PCBs in the stack gases (chlorophenols > chlorobenzenes > PCDFs > PCDDs > PCBs) suggest that they are not likely important participants leading to the production of PCDDs/PCDFs.

In their earlier review of formation processes Lustenhouwer *et al.* (1980) rule out the presence of PCDDs/PCDFs as trace components in refuse and devote the majority of their attention to the contribution of several common industrial/ commercial precursors in the refuse to the direct or indirect formation of PCDDs/PCDFs. These include polyvinyl chloride, polychlorinated benzenes, polychlorinated phenols and biphenyls (PCBs) and 2,4,5-T derivatives. Although they do not discuss de novo synthesis in any detail, they do point out that inorganic chloride is present in municipal refuse and other fuels, and in spite of the fact there is no direct evidence on this formation process it cannot be ruled out as an important contributor.

Recent reports by Ballschmiter et al. (1983) and Tsang and Shaub (1982) have attempted to relate the chemistry of PCDD/PCDF formation to incineration operational parameters. In the former the involvement of chloroethenes and chloroethines in pyrolytic reactions leading to PCDD/PCDF precursors (polychlorobenzenes and polychlorobiphenyls) is detailed and an attempt is made to relate stack gas concentrations of reaction byproducts such as HCl, SO_2, CO_2, H_2O and O_2 to PCDD/PCDF levels. Although no simple correlations were apparent, the authors conclude that municipal incinerators can, in practice, be operated under conditions that lead to low level organic emissions.

The study by Tsang and Shaub (1982) examines the basic combustion process, depolymerization and condensation into char under pyrolytic conditions and oxidation in gas and condensed phases during combustion. They conclude that since organic pollutants are the result of incomplete combustion, modification of the combustion process represents a more feasible abatement approach than the use of additional clean-up devices. The lower heating values and higher ash levels of municipal refuse (compared to coal) as well as the requirement for greater throughputs of air also must be taken into account in the design of effective precipitation devices. Destruction efficiencies (temperature vs. residence time) for a number of hazardous chemicals are presented, and the difficulty in temperature regulation under real operating conditions is underscored. As evidence of the extreme variability in temperature within the combustion process, they report the detection of long chain hydrocarbons, organic acids and plasticizers in emissions from municipal incinerators suggesting that some of the fuel was only heated at low temperature.

The effect of temperature as well as other combustion variables also has been recently examined by Benfenati _et al._ (1983) in a representative 9-month sampling from an urban Italian incinerator. The only significant correlation found was between PCDD or PCDF concentrations (log transformed) and minimal combustion temperature (320-1010°C) suggesting that the lowest temperature reached is the main factor affecting the amount of PCDD or PCDF emitted. Other factors which were not statistically significant included average combustion temperature, stack gas temperature, stack flow and stack gas concentrations of HCl, NO_x, SO_2 and CO.

In Ontario the results of an evaluation of incinerator operating parameters and their relationship to PCDD/PCDF emissions also has been completed (Envirocon Ltd., 1984). A direct correlation between PCDD and PCDF concentrations and the parameters examined, including a limited range of furnace top temperatures (508-741°C), overfire air port flow, total air, total hydrocarbon and carbon monoxide concentrations was not found.

Clearly, as pointed out by Taylor et al. (1983), the delineation of the major mechanisms of PCDD/PCDF formation in combustion environments requires more extensive sampling techniques, coupled with analyses not only of PCDD/PCDF concentrations but also of the various precursor compounds present in the combustion feed and effluents.

These future research concepts are further addressed in the work of Shaub and Tsang (1983) who utilized computer simulation to study the possibility of PCDD formation under typical incinerator combustion conditions. Some of the formation possibilities examined included

1) presence of PCDDs in input feed,
2) the survival of unburned chloro organics through the main combustion chamber into post combustion zones,
3) temperature fluctuations resulting in gas entrapment and PCDD formation above ambient but below flame temperatures,
4) the conversion of molecules other than chlorophenols to PCDDs in the gas phase,
5) inorganic chlorine sources in the fuel resulting in chlorination of organics and gas phase formation of PCDDs, and
6) non gas-phase or combined gas-phase and non-gas-phase reactions leading to the formation of PCDDs.

With regard to gas phase processes it was suggested that PCDD formation is likely to be low provided precursor concentrations also are low. However, the possibility of PCDF formation from PCDDs in the gas phase is highly probable. The non-gas-phase formation of PCDDs also is discussed and it is suggested that reactions involving suspended fly ash or reactive processes within or on the grate beds of incinerators should be considered. One of the author's main conclusions is that although more research is needed it is important to recognize that "there is no thermal or kinetic stability attributable to these compounds that would prohibit their efficient destruction at high temperatures".

Global Emission Data

In an effort to correlate the vast amount of information which has now been published on the release of PCDDs and PCDFs from the incineration of raw and treated municipal waste the global data have been tabularized and assessed for emission trends. This information has been partitioned as follows:

TYPE	DESCRIPTION	ANALYTICAL UNITS
Precipitated fly ash	particulate material collected from the incinerator off-gases by pollution abatement equipment	ng/g
Stack-collected particulates	particulate material that escapes the pollution abatement equipment but is trapped by the stack sampling particulate filter	ng/g
Stack-collected particulates	(as above)	ng/m^3
Stack-collected gas or vapour phase	gaseous or aerosol components that pass through stack sampling particulate filters and are trapped via impinger solutions and cartridges containing sorbent materials such as florisil and XAD-2	ng/m^3
total stack emissions	combined stack-collected particulate plus gas phase emissions (total collection train results)	ng/m^3

In the case of in-stack sampling it must be emphasized at the outset that the partitioning of the published emission data into particulate and gaseous phases may not accurately depict actual atmospheric partitioning of the escaping contaminants due to several artifacts associated with the sampling process. These include gaseous adsorption, particulate off-gassing and more importantly stack gas temperature at the collection point. These factors all have a pronounced but as yet unquantified influence on the physical partitioning process and on isomer group distribution and have not been well addressed in the published accounts which have been cited. Accordingly, the evaluation of the global data base for emission trends and isomer group distribution patterns must be considered in the light of these limitations and should therefore serve only as a very approximate guide for emission characteristics.

In Tables 4.2.1.1A to 4.2.1.1C all available information concerning PCDD and PCDF concentrations in precipitated fly ash and stack collected particulate matter (weight:weight basis) and in stack emissions (weight:volume basis) for particulate, gaseous and total stack emissions are presented. Included in these tables are the locations of the sources (where identified) and the authors of the respective reports. A summary of the emission data is presented in Table 4.2.1.1D. It was apparent from this summary that great variability (standard deviation values greater than corresponding means) existed within each of the individual PCDD and PCDF chlorinated congener series. In view of this variability and

Page 1 of 3

TABLE 4.2.1.1A

LEVELS OF PCDD/PCDF IN PRECIPITATED FLY ASH FROM MUNICIPAL WASTE INCINERATOR PLANTS (ng/g)

Reference	Location	Sampling Interval	PCDD						PCDF						Total PCDD + PCDF
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
Buser & Bosshardt (1978)	Switzerland (Zurich)	1 plant	2	8	30	60	120	220	1	4	30	40	10	85	305
Eiceman et al. (1979),(1980)	Canada (Ontario)	2 plants	12.0 9.3	15	13	3	0.4	41							
	Japan	2 plants	4.8 8.5												
	The Netherlands	1 plant	2.4												
Cavallaro et al. (1980)	Italy	4 plants	6.1 1.1 0.1 2.7				150 36 0.7 166						166 22 0.2 70		
Lamparski & Westrick (1980)	U.S.A.	1 plant	7.8		14	28	30	(72)*							
	Europe	1 plant	69.7		550	1040	650	(2310)*							
Lustenhauwer et al. (1980)	The Netherlands	1 plant - Ave. of 1/wk for 4 wks	58	192	375	363	135	1123	94	192	336	218	22	862	1985
Eiceman et al. (1981)	Canada (Ontario)	1 plant Ave. of 1/wk for 8 wks	13.5	23.2	25.8	14.9	6.3	84							
Little (1981)	Germany	1 plant	25												
Cavallaro et al. (1982)	Italy	5 plants	0.25 0 46.4 0.7 0	1.7 0.92 65.4 0.05 0	294 1.8 2496 0.02 0	8.9 3.1 87.9 0.007 0.0012	295 1.5 841.5 0.1 5.86	600 7 3537 1 6	0.46 0.8 61.7 1.88 0				15.8 3.3 255 0.002 1.93		

TABLE 4.2.1.1A

LEVELS OF PCDD/PCDF IN PRECIPITATED FLY ASH FROM MUNICIPAL WASTE INCINERATOR PLANTS (ng/g)

Reference	Location	Sampling Interval	PCDD						PCDF						Total PCDD + PCDF
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
Clement & Karasek (1982)	Canada (Ontario)	1 plant-Ave. of 2 days	5.0	4.9	4.2	2.5	5.3	22							
Liberti et al. (1982)	Italy	4 plants	2.7	7.4	21	47.1	92	170	10.4	29.5	37	50.5	15	142.4	313
			85	165	595	835	520	2200	285	415	910	715	125	2450	4650
			49	33	14	6	2.7	105	576	216	57	13	15	877	982
			4	108	4	31	5	152	247	358	93	34	3	735	887
Olie et al. (1982)	The Netherlands	9 plants	101		310		40		230		230		0		
			13		250		1370		90		230		90		
			24		136		51		91		82		11		
			226		560		110		240		280		10		
			212		910		550		220		530		110		
			0		10		10		50		60		10		
			40		330		190		110		150		40		
			18		140		190		70		70		20		
			373		1340		990		430		1200		140		
Ballschmiter et al. (1983)	W. Germany	6 plants					0.5								
							1								
							13								
							15								
							520								
Clement et al. (1983)	Canada (Ontario)	1 plant Ave. of 8 days	26	29	20	7	13	95							
	Canada (Ontario)	1 plant Ave. of 1/hr over 5 hrs.	28	30	22	8	14	102							
Karasek & Viau (1983)	W. Germany	1 plant Ave. of 3 mos.	0	0	0.3	1.2	11.8	13	9.9	2.9	3	3.8	4.9	24.5	38
Olie et al. (1983)	The Netherlands (Alkmaar)	1 plant Ave. over 17 mo.	114	205	435	247	96	1097	220	304	421	206	18	1169	2266

Page 3 of 3

TABLE 4.2.1.1A

LEVELS OF PCDD/PCDF IN PRECIPITATED FLY ASH FROM MUNICIPAL WASTE INCINERATOR PLANTS (ng/g)

Reference	Location	Sampling Interval	PCDD						PCDF						Total PCDD + PCDF
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
Olie et al. (1983) (Cont'd)	The Netherlands (Amsterdam)	1 plant Ave. over 18 mo.	14	39	126	202	401	782	62	62	128	82	33	367	1149
	The Netherlands (Zaanstad)	1 plant Ave. over 2.5 hr.	89	434	1522	1702	974	4721	183	399	863	542	94	2081	6801
	The Netherlands (Zaanstad)	1 plant Ave. over 21 mo.	107	310	731	687	359	2194	211	367	590	395	60	1623	3817
Ozvacic et al. (1984a)	Canada (Ontario)	1 plant 13 tests over 2 mos.	3.7	6.4	9.1	2.3	1.5	23	12	17	14	2.9	0.3	46	69
Ozvacic (1985)	Canada (Ontario)	2 plants Ave. of 3 tests over 2 wks.	2120 30	1924 34	7124 80	2980 25	1015 9	15163 178	663 53	1007 59	2285 112	1178 21	118 7	5251 252	20414 430

* Excludes P_5CDD

TABLE 4.2.1.1B

LEVELS OF PCDD/PCDF IN STACK-COLLECTED PARTICULATES FROM MUNICIPAL WASTE INCINERATOR PLANTS(ng/g)

Reference	Location	PCDD						PCDF						Total PCDD + PCDF
		T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
Lustenhouwer *et al.* (1982)	The Netherlands)	100	800	1370	1370	310	3950	460	960	1600	1130	140	4290	8240
Little (1981)	Germany	300												
Liberti & Brocco (1982)	Italy (Florence)		80	180	290	510	(1060)*				110	60		1230
Chiu *et al.* (1983)	Canada	10 0	269 168	390 173	11 127	8 1	688 469	46 279	153 265	1712 392	47 104	0 0	1958 1040	2646 1509
Ozvacic *et al.* (1984a)	Canada (Ontario)	2100	2300	2900	1700	1900	11000	8500	8000	4600	1400	370	23000	34000

* Excludes T_4CDD

TABLE 4.2.1.1C

Page 1 of 3

LEVELS OF PCDD/PCDF IN STACK EMISSIONS FROM MUNICIPAL WASTE INCINERATORS (ng/Nm^3 or dscm)

Reference	Location	Sample Type*	PCDD						PCDF						Total PCDD + PCDF	** Units
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total		
Cavallaro *et al.* (1980)	Italy (Zanu - Milan)	P	3				290						37			ng/Nm^3
		G	19				732						675			
		T	22				952						712			
	Italy (Busto Arsizio)	P	0.6				37						70			
		G	0.8				11						10			
		T	1.4				48						80			
Ahling & Lindskog (1982)	Sweden	T (3 temp. regimes)			5	13	20									ng/Nm^3 NTP
					100	800	1000									
					40	30	300									
Cavallaro *et al.* (1982)	Italy (6 plants)	P	1.1	2.7	11.5	1.0	8.0	24.3	ND				2.2			ng/Nm^3
		G	19.6	27.9	178.2	159.6	63.9	449.2	ND				59.3			
		T	20.7	30.6	189.7	160.6	71.9	473.5	ND				61.5			
		P	172.2	172.3	12015	575	7312	20247	75.0				2883			ng/Nm^3
		G	17.0	107.0	26620	828	1179	28751	108.6				4390			
		T	189.2	279.3	38635	1408	8491	49003	183.6				7273			
		P	0.04	0.3	6.7	0.2	1.7	8.9	2.6				0.1			ng/Nm^3
		G	19.0	40.0	6542	124.0	776.0	7501.0	429.0				1010.0			
		T	19.0	40.3	6549	124.2	777.7	7509.9	431.6				1010.1			

TABLE 4.2.1.1C

Page 2 of 3

LEVELS OF PCDD/PCDF IN STACK EMISSIONS FROM MUNICIPAL WASTE INCINERATORS (ng/Nm^3 or dscm)

Reference	Location	Sample Type*	PCDD						PCDF						Total PCDD + PCDF	** Units
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total		
Cavallaro et al. (1982) (Cont'd)	Italy (6 plants) (Cont'd)	P	10.3	2.8	0.5	3.2	39	56.4	3.7				0.1			ng/Nm^3
		G	60.0	33.0	1390.0	167.0	2703	4353.0	1814.0				1760.0			
		T	70.9	35.8	1390.5	170.2	2742	4409.4	1817.7				1760.1			
		P	0.3	2.4	196	9.9	173	381.6	75.3				3.2			ng/Nm^3
		G	9.6	21.0	328	46.0	244	648.6	305.0				89.0			
		T	9.9	23.4	524	55.9	417	1030.2	380.3				92.2			
		P	ND	0.01	0.3	ND	0.5	0.8	ND				ND			ng/Nm^3
		G	19.0	11.0	480.0	6.0	71.0	587.0	27.0				24.0			
		T	19.0	11.01	480.3	6.0	71.5	587.8	27.0				24.0			
Gizzi et al. (1982)	Italy (como)	T- Ave. over 9 months	130	199	366	286	126	1107	309	250	314	215	124	1212	2319	ng/Nm^3
Liberti & Brocco (1982)	Italy (Florence)	P													1570	ng/m^3
		G													261	
Olie et al. (1982)	The Netherlands (Zaanstad)	P- Ave. over 7 months	57	231	440	347	452	1527	161	272	528	293	68	1322	2849	ng/Nm^3
SFOEP (1982)	Switzerland (Zurich - Josef-strasse)	P	3.4	9.9	22.7	22.1	47.5	106	15.5	19.9	14.3	9.7	7.7	67	173	ng/Nm^3
		G	0.6	1.1	2.2	2.0	1.1	7	66.8	7.4	4.4	2.7	0.5	82	89	
		T	4.0	11.0	24.9	24.1	48.6	113	82.3	27.3	18.7	12.4	8.2	149	262	
Olie et al. (1983)	The Netherlands	T	25	85	231	194	166	701	65	141	242	161	84	693	1394	ng/Nm^3

Page 3 of 3

TABLE 4.2.1.1C

LEVELS OF PCDD/PCDF IN STACK EMISSIONS FROM MUNICIPAL WASTE INCINERATORS (ng/Nm^3 or dscm)

Reference	Location	Sample Type*	PCDD						PCDF						Total PCDD + PCDF	** Units
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total		
Orvacic (1983)	Canada (Ontario)	T- Ave. of 3 tests	61	88	392	437	104	1082	226	177	365	245	72	1085	2167	ng/dscm
	Canada (Ontario)	T- Ave. of 3 tests	45	53	75	423	144	740	342	558	71	104	138	1213	1953	ng/dscm
	Canada (Ontario)	T- Ave. of 3 tests	4658	2543	454	319	67	8041	13231	6932	104	43	47	20357	28398	ng/dscm
Rappe et al. (1983a)	Sweden (Eksjo)	P	ND	ND	ND	20	13	33	37	10	25	25	9	106	139	ng/m^3
		G	ND	ND	ND	7	4	11	68	6	10	7	1	92	103	
		T	ND	ND	ND	27	17	44	105	16	35	32	10	198	242	
		P	ND	ND	ND	12	10	22	25	4	4	8	8	49	71	ng/m^3
		G	ND	ND	ND	25	8	56	525	87	80	15	6	713	769	
		T	ND	ND	ND	37	18	78	550	91	84	23	14	762	840	
Redford et al. (1983)	U.S.A.	T	6.3	NR	16	7.6	2.5	32.4 ***	90	NR	62	7.5	0.6	160.1 ***	192.5 ***	ng/dscm
Tiernan et al. (1983b)	U.S.A.	T	380	530	850	2000	490	4300	2600	1600	1800	2200	170	8300	12600	ng/m^3
Orvacic et al. (1984a)	Canada (Ontario)	P (Ave.	387	361	351	153	117	1369	1257	1110	520	103	29	3020	4389	ng/dscm
		G of 13	373	349	339	147	113	1321	1303	1150	540	107	31	3130	4451	
		T tests)	760	710	690	300	230	2690	2560	2260	1060	210	60	6150	8840	

P - particulate
G - gaseous
T - Total of particulates and gaseous data.
** 1 Nm^3 (normal cubic metre) = 0.925 dscm (dry standard cubic metre).
*** - Total PCDD/PCDF value excludes P_5CDD and P_5CDF.

considering that comparatively few studies were conducted by analyzing for all five chlorinated PCDD/PCDF isomer groups (T_4 to O_8) the average PCDD and PCDF congener values for all studies were summed (Table 4.2.1.1D) to yield total (T_4 to O_8) PCDD and PCDF values.

In order to further assess the relative contribution of each chlorinated PCDD or PCDF isomer group to the total, a separate analysis of the data in Tables 4.2.1.1A and 4.2.1.1C was conducted using only those studies in which all five PCDD or PCDF isomer groups were reported. These percentage distribution data are presented in Table 4.2.1.1E.

Precipitated Fly Ash

From the global data in summary Table 4.2.1.1D (subject to the caveats described above) it is apparent that average total (T_4 to O_8) PCDD and PCDF concentrations in precipitated fly ash each are in the 1 ug/g range with an average total PCDD plus PCDF concentration of just over 2 ug/g.

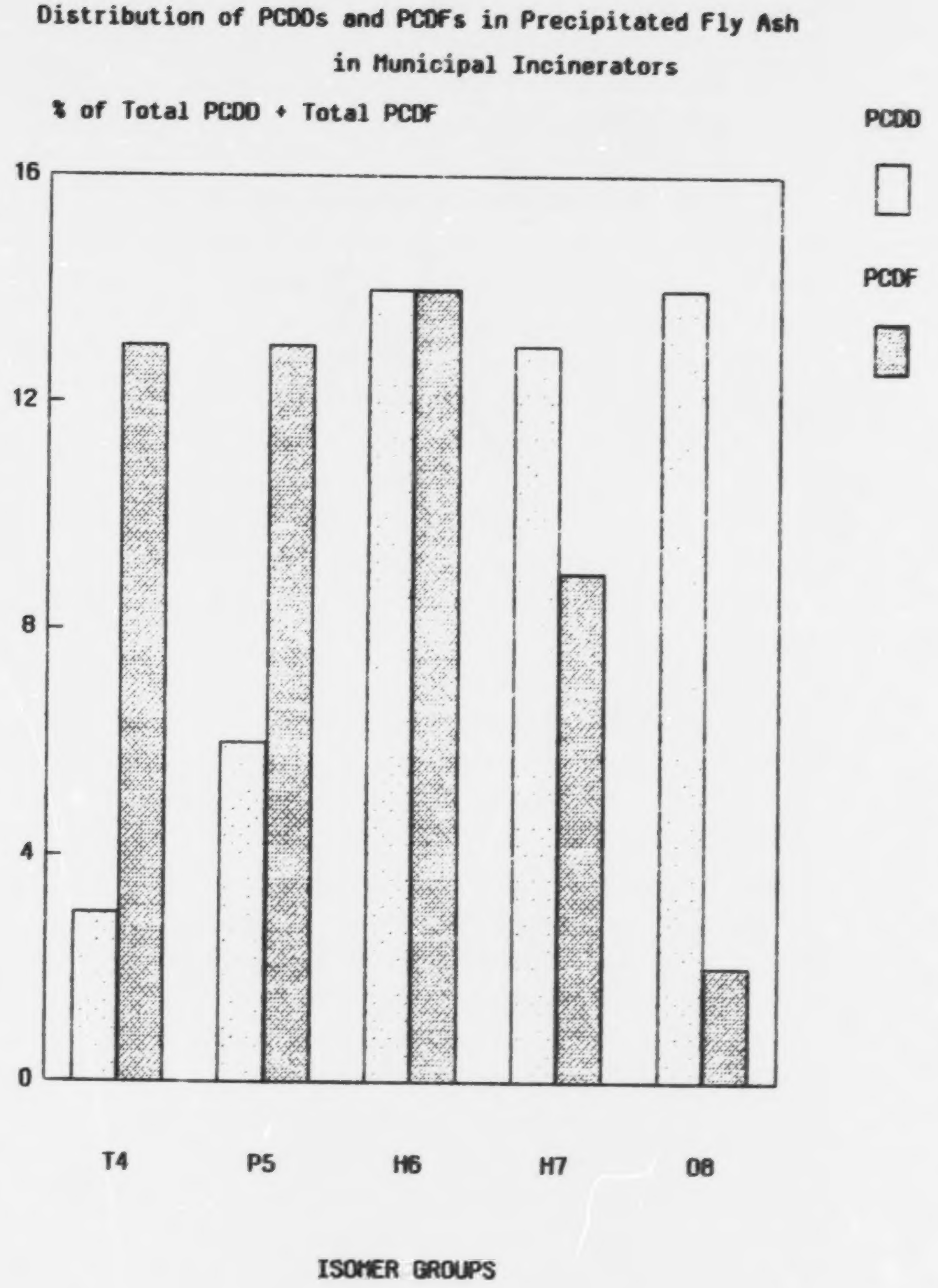
Within the PCDDs (Table 4.2.1.1E) the H_6 and O_8 isomer groups are present in greatest quantities (28 and 23%, respectively) while the H_7, P_5 and T_4 isomer groups comprise the remainder at levels of 19, 17 and 13%, respectively. In contrast, the T_4 - H_7 CDFs are present in approximately similar proportions (20-30%) with O_8CDF comprising a much lower 5%. In studies where both PCDDs and PCDFs were reported, the average percentage composition of each chlorinated isomer group, compared to total PCDD plus PCDF, ranged from lows of 2% (T_4CDD and

T_4CDF) to highs of 13 to 14% (H_6- to O_8CDD and T_4- to H_6CDF), with the total PCDDs and PCDFs present in about a 50:50 distribution. A graphic depiction of the percentage distribution of PCDDs and PCDFs in fly ash is shown in figure 4.2.1.1A.

In an effort to assess the variation in PCDD and PCDF congener concentrations all studies in which samples were collected from a given source over a number of hours, days, weeks or months were utilized in the derivation of statistical coefficients of variation (Table 4.2.1.1F). In the case of fly ash it is apparent that variation in PCDD and PCDF isomer group concentrations over a sampling period as short as 2 to 5 hours can range from 18 to 45% while over weeks or months, this variation can range from less than 10% to well in excess of 100%.

Samples of fly ash also have been analyzed for PCDDs on a size-fractionated basis (Table 4.2.1.1G). In the case of an Ontario source very little difference in total (T_4 to 0_8) PCDD concentration was detected in sieve-separated fly ash particles ranging in size from 30-850 um. However, in a similar study Karasek et al. (1982) found much higher total (T_4 to 0_8) PCDD concentrations in 30 um particles from a French incinerator with progressively lower levels on particles of increasing size. Caution must be used in evaluating these findings as the mechanical separation may have affected the particle size distribution. More information on this phenomenon via size-fractioned in-stack sampling is required.

Figure 4.2.1.1.A



* Data from Table 4.2.1.1 E

TABLE 4.2.1.1D

SUMMARY[1] OF AVERAGE PCDD AND PCDF CONGENER CONCENTRATIONS AND CALCULATED TOTAL (T4-O8) VALUES FOR MUNICIPAL WASTE INCINERATOR EMISSIONS

Matrix	Units**	PCDD						PCDF						Total* (T_4-O_8) PCDD + PCDF
		T4	P5	H6	H7	O8	Total* (T_4-O_8)	T4	P5	H6	H7	O8	Total* (T_4-O_8)	
						- Weight Basis -								
Precipitated Fly Ash	ng/g Ave.	90	151	528	322	227	1318	151	245	379	250	47	1072	2390
	n	44	24	35	26	45		28	14	23	14	32		
	s.d.	321	394	1265	680	344		172	271	527	348	61		
Stack-Collected Particulates	ng/g Ave.	502	723	1003	700	546	3474	2321	2345	2076	558	114	7414	10888
	n	5	5	5	5	5		4	4	4	5	5		
	s.d.	901	925	1170	778	787		4123	3787	1786	653	154		
						- Volume Basis*** -								
Stack-Collected Particulates	ng/m^3 Ave.	49	71	1186	104	654	2064	150	283	218	88	240	979	3043
	n	13	11	11	11	13		11	5	5	5	13		
	s.d.	112	126	3595	189	2005		370	476	279	121	795		
Stack-Collected Gaseous Phase	ng/m^3 Ave.	45	59	3588	151	492	4335	465	313	159	33	671	1641	5976
	n	12	10	10	10	12		10	4	4	4	12		
	s.d.	104	107	8334	247	798		615	560	257	50	1295		
Total Stack Emissions		338	290	2551	343	741	4263	1353	1205	378	296	618	3850	8113
	ng/m^3 n	19	16	20	20	22		17	10	11	11	19		
	s.d.	1062	635	8614	516	1835		3177	2151	558	638	1673		

---

[1] - from Tables 4.2.1.1A to 4.2.1.1C

* Total - Calculated from sum of average isomer group values.

** n. = number of values averaged.
s.d. = standard deviation.

*** - data for Nm^3 and dscm averaged without correction (1Nm^3 = 0.925 dscm).

**** - total of gaseous plus particulate phases.

TABLE 4.2.1.1E

AVERAGE PERCENT COMPOSITION * OF PCDDs AND PCDFs IN EMISSIONS FROM MUNICIPAL WASTE INCINERATORS

Matrix	Units	PCDD						PCDF					
		T_4	P_5	H_6	H_7	O_8	Total (T_4-O_8)	T_4	P_5	H_6	H_7	O_8	Total (T_4-O_8)
Precipitated Fly ash	% of PCDD or PCDF	13	17	28	19	23		21	23	32	21	5	
	% of PCDD + PCDF	3	6	14	13	14	49	13	13	14	9	2	51
Stack-Collected Particulates	% of PCDD or PCDF	6	7	31	17	39		33	21	22	16	8	
	% of PCDD + PCDF	3	4	7	12	14	40	21	12	13	9	5	60
Stack-Collected Gaseous Phase	% of PCDD or PCDF	9	6	44	20	21		68	16	11	4	1	
	% of PCDD + PCDF	3	2	3	4	2	14	59	13	9	4	1	86
Total Stack Emissions	% of PCDD or PCDF	10	9	35	24	22		40	23	18	13	6	
	% of PCDD + PCDF	5	5	8	11	6	35	28	15	11	8	3	65

* Percent compositon averages calculated from reported emission data where complete congener (T_4-O_8) levels for PCDD and/or PCDF were reported.

TABLE 4.2.1.1F

VARIATION IN PCDD AND PCDF CONCENTRATION IN PRECIPITATED FLY ASH, STACK COLLECTED PARTICULATES AND TOTAL STACK EMISSIONS FROM SAMPLES COLLECTED OVER DIFFERENT TIME INTERVALS FROM MUNICIPAL WASTE INCINERATORS

Reference	Location	Sample Interval	PCDD						PCDF						Total PCDD+ PCDF
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
			-Coefficient of Variation (%)-												
Lustenhouwer et al., (1980)	The Netherlands	Fly Ash(ng/g) 1/wk for 4 wks	29	21	9	21	47	11	47	19	11	23	27	13	9
Eiceman et al. (1981)	Canada (Ontario)	Fly Ash(ng/g) 1/wk for 8 wks	16	57	62	87	132	61							
Gizzi et al. (1982)	Italy (Como)	Total Stack Emissions (ng/m^3) 17 samples over 9 months	234	259	258	250	133	238	234	222	232	168	152	205	217
Olie et al. (1982)	The Netherlands (Zaanstad)	Stack Particulate(ng/m^3) a - 14 samples over 9 months b - 8 samples over 5 months	116^a	100^b	109^a	73^b	212^a	85^b	90^a	81^b	78^a	43^b	88^a	74^b	78^b
	The Netherlands (Zaanstad)	Fly Ash(ng/g) 8 samples over 2.5 hours	27	30	21	18	32	18	35	29	28	20	45	1	17

TABLE 4.2.1.1F

VARIATION IN PCDD AND PCDF CONCENTRATION IN PRECIPITATED FLY ASH, STACK-COLLECTED PARTICULATES AND TOTAL STACK EMISSIONS FROM SAMPLES COLLECTED OVER DIFFERENT TIME INTERVALS FROM MUNICIPAL WASTE INCINERATORS

Reference	Location	Sample Interval	PCDD						PCDF						Total PCDD+ PCDF
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
			Coefficient of Variation (%)												
Karasek & Viau (1983)	W. Germany	Fly Ash(ng/g) 3 samples over 3 months	0	0	33	92	60	61	158	152	133	142	161	154	121
Olie et al. (1983)	The Netherlands (Alkmaar)	Fly Ash(ng/g) a - 17 samples over 17 months b - 4 samples over 3 months	40^a	31^b	44^a	54^b	70^a	40^b	37^a	31^b	65^a	49^b	167^a	29^b	34^b
	The Netherlands (Amsterdam)	Fly Ash(ng/g) a - 23 samples over 21 months b - 5 samples over 4 months	86^a	100^b	73^a	31^b	107^a	20^b	71^a	60^b	59^a	9^b	97^a	30^b	21^b
	The Netherlands (Zaanstad)	Fly Ash(ng/g) a - 23 samples over 21 months b - 9 samples over 5 months	67^a	51^b	57^a	66^b	84^a	64^b	54^a	63^b	55^a	65^b	103^a	59^b	60^b

Another factor which has been confirmed in the presentation of the global fly ash data in Table 4.2.1.1A is the significantly lower PCDD and PCDF concentration (ng/g) in precipitated fly ash from Ontario waste incineration sources compared to similar sources from other countries. The only exception to that general finding was much higher PCDD and PCDF levels in precipitated fly ash from the Commissioner St. refuse incinerator in Toronto (Ozvacic, 1985).

Stack Emissions

a) Particulates (weight:weight basis)

Fewer studies have been conducted to determine the PCDD/PCDF concentration (weight:weight basis) in stack collected particulate matter (particulates which escape collection by pollution abatement devices) compared to precipitated fly ash. From summary Table 4.2.1.1D it is apparent that **total (T_4 to O_8) PCDDs and PCDFs are present in concentrations approximately twice and seven times higher than those of the fly ash analyses.** The lower result contrasts with the value of ten times which was reported by Lustenhouwer et al. (1980) who were working with an even smaller data base.

However the most recent work on an Ontario refuse incinerator (SWARU, Hamilton) (Ozvacic et al., 1984a) generates even greater uncertainty in terms of the ratio of emitted : precipitated particulate PCDD and PCDF concentrations. Although the stack-collected particulate concentrations (ng/g) in this study were not directly reported they are included in included in Table 4.2.1.1B. The ratios (emitted:precipitated) have been calculated from these data with PCDDs and PCDFs in the stack-collected particulates being about 490 and 550 times higher, respectively, than the corresponding PCDD and PCDF precipitated fly ash results (Table 4.2.1.1A - Ozvacic et al., 1984a).

b) Particulates, Gas phase and Total Stack Emissions (weight:volume basis)

The stack emission data (Table 4.2.1.1C) have, where possible, been subdivided into particulate and gaseous phases and are expressed in ng/m^3 in Summary Table 4.2.1.1D. No adjustment has been made to correct the raw data for minor differences between dry standard (dscm) or normal (Nm^3) gas conditions, as reported. Further, it must be re-emphasized that the partitioning of the published emission data into particulate and gaseous phases may not accurately depict actual atmospheric partitioning of the escaping contaminants due to various sampling artifacts described in the introductory statement.

Subject to these limitations average total stack emissions (Table 4.2.1.1D) are 4 ug/m^3 each for T_4 - O_8 PCDD and PCDF. The corresponding T_4 - O_8 PCDD and PCDF values for particulate and gaseous component emissions were 2 and 1 ug/m^3 (particulates) and 4 and 2 ug/m^3 (gaseous), respectively. As stated earlier the total PCDD and PCDF (T_4 - O_8) values in Table 4.2.1.1D were calculated by summing the average isomer group values for each of the five isomer groups.

It should be noted here that the sum of the PCDDs and PCDFs for the two components does not equal the respective total stack emission numbers. The reason for this apparent anomaly is the fact that

TABLE 4.2.1.1G

SIZE FRACTIONATED PCDD CONCENTRATION IN FLY ASH FROM MUNICIPAL WASTE INCINERATORS

Reference	Location/ Source Description	Matrix	Size (um)	Size Fractionated PCDD Concentrations (ng/g)					
				T4	P5	H6	H7	O8	Total (4-8)
Clement & Karasek (1982)	Canadian (Ontario) Municipal Waste Incinerators	Fly Ash (Average of two days collection)	30	2.6	3	3	2.5	7.3	20
			80	3.2	3.5	3.5	2	3.7	15
			125	5.7	6.5	6	3	5.9	30
			200	7.6	8	7.5	3.5	5.1	30
			550	13.5	10	6	2	1.9	35
			>850	7.9	4.5	1.5	0.2	0.2	15
			>850	3.2	4	3.5	2.5	9.6	25
Karasek *et al.*, (1982)	French Municipal Waste Incinerator	Fly Ash	30	ND	5.0	8	30	120	160
			85	ND	0.8	8	20	39	67
			125	ND	1	4	8	12	25
			200	ND	0.6	1	ND	4.1	6
			550	ND	0.4	0.8	0.4	2.3	4
			>850	ND	ND	ND	4	8.4	13

ND - Not detected

the data base (Table 4.2.1.1C) consists of several studies where only gaseous or particulate values were recorded. Accordingly, these numbers were utilized in the derivation of the summary data for gaseous and particulate components or expressed in 4.2.1.1D but could not be used in the workup for total stack emissions. The total stack emission averages were calculated independently using only those studies where both components were reported or where a total sampling train result was published.

In view of the apparent greater contribution from the gaseous component of the stack emissions a more detailed analysis of the data in Table 4.2.1.1C was performed. This involved the calculation of the percentage composition of the total stack emissions using particulate and gaseous data pairs from only those studies in which both were reported. This analysis revealed that **particulate-borne PCDD and PCDF emissions comprised 31% and 22% respectively, of total stack emissions with the remaining 69% and 78% respectively, being emitted in a non-particulate form.** These ratios differ slightly from those in Summary Table 4.2.1.1D. However, as they were derived using only those studies in which gaseous and particulate data pairs were reported and were further restricted to studies in which all T_4-O_8 PCDD and PCDF isomer groups were reported, they are believed to depict more accurately the relationship between the two emission phases.

An examination of the percent distribution data (Table 4.2.1.1E) revealed several other phenomena which are important in the evaluation of incinerator emissions. On a total particulate plus gaseous stack emission basis (including only those studies where both particulate and gaseous phases - either separately or as a total - were reported) it is apparent that PCDDs comprise about 33% of total emissions with the remaining 67% present as PCDFs.

On a isomer group distribution basis the composition of total PCDD plus PCDF emissions ranges from lows of about 3% (T_4CDD and O_8CDF) to a high of about 30% (T_4CDF). The average percentage composition within the PCDDs and PCDFs is shown in Table 4.2.1.1E. PCDDs range from lows of 7.5% and 7.1% for T_4- and P_5CDD to a high of 38.8% for H_6CDD and from a low of 5.4% (O_8CDF) to a high of 41.2% (T_4CDF) for the PCDFs.

Although the particulate component of stack emissions appears to follow the same isomer group distribution pattern as the total stack emissions (Table 4.2.1.1E), the gaseous phase results differ significantly, with PCDDs comprising only 14% of total PCDD plus PCDF emissions. This difference appears to be related to a much greater contribution (59%) from T_4CDF. As this difference does not appear in Table 4.2.1.1D where all data were included in the averaging process, it may reflect the use of a limited number of studies (4) where all five PCDD and PCDF isomer groups were reported for the gaseous component of the emissions. Variability in stack temperature also may account for this distribution pattern.

As in the case of the precipitated fly ash and stack-collected particulate matter PCDD and PCDF data (weight:weight basis) the variability in the stack emission data (weight:volume basis) is fairly high (Table 4.2.1.1D) and renders the isomer group distribution pattern generalizations (Table 4.2.1.1E) from the foregoing description of only limited value. However, as with the fly ash data, there were two studies where stack emission data were taken on a replicated basis (up to 9 months) and this has permitted statistical coefficient of variation analysis (Table 4.2.1.1F).

In contrast to the coefficient of variation results for PCDD and PCDF isomer group concentrations in precipitated fly ash which ranged from 0-167% with most in the 11-70% range, the corresponding variation in the stack-collected emission concentrations at the plants in Italy and the Netherlands was much higher, ranging from 43-259%, with most over 100%.

Isomer Specific Analyses

Although some progress has been made in isomer specific analysis of emissions from municipal waste incinerators since the publication of the 1981 NRCC document, there are still many deficiencies in the data base. Tables 4.2.1.1H to 4.2.1.1J summarize this information.

From Table 4.2.1.1H it is apparent that between 50 to 60% of the total number of possible isomers in the T_4, P_5 and H_6 CDD and CDF congener series usually are detected. Usually all of the H_7 and O_8 CDD and CDF isomers are detected. On average, a total of 30 and 40 isomers of PCDD and PCDF respectively can be expected in precipitated fly ash and stack emissions.

From a quantitative perspective the information published to date involves isomers in the T_4CDD isomer series. These data are presented in Table 4.2.1.1I and are summarized in Table 4.2.1.1J. The major T_4 isomers are 1,3,6,8- and 1,3,7,9-T_4CDD. The 1,3,7,8-, 1,2,4,6- and 1,2,4,9-T_4CDD isomers also are present in significant proportions of about 10%, which is approximately two times higher than if the T_4CDD isomers were equally distributed. The most toxic isomer 2,3,7,8-T_4CDD can be expected in a range from about 3-7% (average 5%) of the total T_4CDD content of incinerator emissions.

On the basis of these data, it is apparent that isomers within the T_4CDD series are not distributed equally. Although there is a paucity of published information on this subject for other PCDD and PCDF congener series it can be stated, on the basis of personal conversation (R.E. Clement MOE, 1984), that in the case of Ontario studies using EC-MS electron impact the response factors in the same isomer group do not change by more than about a factor of two, but peak areas for observed isomers often differ by more than a factor of 10 for penta and hexa isomer groups. Usually 2 - 3 of the 6 - 14 isomers detected contain more than 50% of the total quantity detected for a specific isomer group.

Atmospheric Dispersion and Annual Output of PCDDs and PCDFs

The few reported cases where dispersion modelling has been applied to stack emission parameters are summarized in Table 4.2.1.1K.

It is apparent from these results that this type of information is of relatively little practical or predictive value due to its limited nature (3 studies), specificity to the emission and operational parameters of the incinerator, the stack height and to the characteristics of the model which was selected based on the atmospheric conditions in the vicinity of the source. A more extensive data base (Table 4.2.1.1L) is, however, available to establish the significance of municipal waste incinerators in terms of their annual PCDD and PCDF output to the atmosphere. Data on emissions from waste incinerators in Ontario have not been included in this table as they are discussed in detail in Section 4.2.1.7.

On the basis of these studies (Table 4.2.1.1L) which involve emissions from incinerators in four different countries it is apparent that average combined PCDD and PCDF annual outputs per incinerator range from 150-4500 g per year. In the Italian study (Gizzi _et_ _al._, 1982) the sampling was conducted over a 9-month period, thereby allowing the derivation of maximum (5918), minimum (58) and average (691) annual PCDD plus PCDF (T_4 to O_8) combined gaseous and particulate outputs (g/year).

In a follow-up study (Benfenati _et_ _al._, 1983), these emissions (gaseous and particulate) were reported on the basis of the weight of refuse incinerated and ranged from 1730 ug to 229,096 ug with a mean of 30,592 ug per ton of refuse combusted. However, a more representative mean excluding two exceptionally high values (resulting from a very low, minimum combustion temperature) has been calculated at 6755 ug per ton or 7.4 ug/kg. This value is about 5 times higher than the

TABLE 4.2.1.1H

NUMBER OF ISOMERS DETECTED IN PCDD AND PCDF CHLORINATED CONGENERS IN EMISSIONS FROM MUNICIPAL WASTE INCINERATORS

Chlorinated Congener Group	Total Isomers Possible	Number of Isomers Detected												Average No. of Isomers Detected
		Redford et al. 1983	Tiernan et al. 1982	Buser et al. 1978 a, b	Tiernan et al. 1983b	Cavallaro et al. 1980		Buser & Rappe 1980	Lamparski & Nestrick 1980		Ozvacic et al. (1984a, b)		Clement et al. (in press)	
		stack particulate plus gaseous phase	stack particulate	stack particulate	stack particulate plus gaseous phase	stack dust/ fly ash	stack gaseous phase	fly ash	U.S.	Europe	stack particulate plus gaseous	fly ash	stack particulate plus gaseous and fly ash	
T4 CDD	22	9	17	8	11	10	9	17	18	20	12	12	11	13
P5 CDD	14			9	8	9	7				12	12	11	10
H6 CDD	10	5		8	8	7	7				7	7	7	7
H7 CDD	2	2		2	1	2	2				2	2	2	2
O8 CDD	1	1		1	1	1	1				1	1	1	1
T4 CDF	38	5		21	17	12	11				15	14	17	14
P5 CDF	28			17	12	15	14				13	12	14	14
H6 CDF	16	5		7	11	11	10				9	9	13	9
H7 CDF	4	1		4	4	4	4				2	4	4	3
O8 CDF	1	1			1	1	1				1	1	1	1

TABLE 4.2.1.11

ISOMER SPECIFIC ANALYSES SHOWING PERCENTAGE OF TOTAL T_4CDD MEASURED IN MUNICIPAL WASTE INCINERATOR EMISSIONS

Isomer or Isomer Grouping	Lamperski & Nestrick (1980)		Tiernan *et al.* (1982a)			Tiernan *et al.* (1983b)	Redford *et al.* (1983)	Cavallaro *et al.* (1980)		
	Fly Ash U.S.	Fly Ash European	Stack Particulate ** 10um+ 3um	Stack Particulate ** 1um	Stack Particulate ** <1um	Stack Particulate	U.S. MRI	Fly Ash	Stack Particulate	Stack Gaseous Phase
	(% of Total T4CDD)									
1,3,6,8-	17.0	23.3	22	17	17	20.97				
1,3,7,9-	13.0	10.1	16	9	6	13.29				
1,3,6,9-			5	5	5	5.17				
1,3,7,8-	17.7	19.0								
1,3,7,8-/1,4,6,9-/1,2,4,8-						10.64				
1,3,7,8-/1,4,6,9-/1,2,4,8-/1,2,4,7-			13	13	13					
1,2,4,7-/1,2,4,8-	4.0	9.9								
1,4,6,9-	<0.6	0.4								
1,2,4,6-/1,2,4,9-	9.4	5.0				14.77				
1,2,6,8-			6	9	8					
1,2,6,8-/1,2,7,9-	6.5	3.6				7.09				
1,2,7,9-			7	8	6	2.07				
1,4,7,8-			0*	0*	0*	6.65				
1,2,3,4-	4.8	3.0								
1,2,6,9-	2.5	1.4								
1,2,3,6-/1,2,3,9-	5.5	3.3								
1,2,3,4/1,2,3,6/1,2,6,9-			16	14	13	11.08				
2,3,7,8-	5.5	3.3					6.5			
2,3,7,8-/1,2,3,8-/1,2,3,4-								3.8,5.8,5.0	4.3,2.0,3.1	9.0,9.3,7.0
1,2,3,7-/1,2,3,8-	9.3	12.2								
1,2,3,7-/1,2,3,8-/2,3,7,8-			13	13	23	3.84				
1,2,7,8-	<1.0	4.5				4.43				
1,2,6,7-			0*	6	6					
1,2,8,9-			0*	3	5					
1,2,6,7-/1,2,8,9-	2.8	1.1								

* - Less than detection limit (not specified).

** - Extracted with less efficient methylene chloride.

calculated emission factor of 1.5 ug/kg refuse burned which has been suggested as the probable PCDD plus PCDF emissions from a proposed new municipal waste to energy plant approved for a site in Ontario (Kemp, 1983).

TABLE 4.2.1.1J

AVERAGE ISOMER COMPOSITION (%) OF TOTAL T_4CDDs IN FLY ASH AND STACK EMISSIONS

ISOMER OR ISOMER GROUP	NO.* ANALYSES	AVE. % OF TOTAL T_4CDD
1,3,6,8-	6	20
1,3,7,9-	6	12
1,3,7,8/1,4,6,9/1,2,4,9/1,2,4,7-	10	10
1,2,4,6/1,2,4,9-	3	10
1,2,6,9/1,2,3,6/1,2,3,9/1,2,3,4-	10	8
2,3,7,8/1,2,3,7/1,2,3,8/1,2,3,4-	18	8
1,2,6,8/1,2,7,9-	10	6
1,2,6,9-	4	5
1,2,7,8-	3	3
1,2,6,7/1,2,8,9-	8	3
1,2,7,8-	4	2

- underlined isomers in groups are those present in highest concentration.

* - for individual references see Table 4.2.1.1I.

TABLE 4.2.1.1K

DISPERSION MODEL PREDICTIONS OF GROUND LEVEL CONCENTRATION IN THE VICINITY OF MUNICIPAL WASTE INCINERATORS

Reference	Source Location/ Description	Model Used	Description of Modeled Parameters and Units	MEASURED STACK GAS CONCENTRATIONS AND CORRESPONDING PREDICTED GROUND LEVEL CONCENTRATIONS													
				PCDD							PCDF						TOTAL PCDD + PCDF
				T_4	P_5	H_6	H_7	O_8	Total	2378	T_4	P_5	H_6	H_7	O_8	Total	Total
SFOEP (1982)	Switzerland-Zurich Josefstrasse - (gas flow rate 94,000 Nm^3/hr)	Gaussian disperson model	Measured total stack gas conc. (ng/Nm^3)	4	11	24.9	24.1	49.1	113.1	NR	22.3	27.3	18.7	12.4	8.2	88.9	202.0
			Predicted max. ave. ground level conc. as aerosol and suspended particulate (pg/m^3)	0.012	0.032	0.075	0.072	0.15	0.341		0.067	0.082	0.056	0.037	0.025	0.267	0.608
			Predicted max. ave. dust deposition (pg/m^2/day)	10	28	65	63	130	296	0.41	58	71	49	32	21	231	527
Barnes (1983)	USA - 5 municipal waste combustors Feed Rates: 1000-17000 kg/hr Comb. Temp = 550-1200°C Gas Flow Rate = 3.7-83.3 dscm/sec Stack ht. = 10-76 m	PTMAX	Measured Stack gas conc. (ng/dscm)	ND-8.5	NR	NR	NR	NR	NR	ND-3.5	NR	NR	NR	NR	NR	NR	NR
			Predicted max. annual ave. ground level conc. (pg/m^3)	up to 0.092	NR	NR	NR	NR	NR	up to 0.038	NR	NR	NR	NR	NR	NR	NR
Olie *et al.* (1983)	The Netherlands Zaanstad Incinerator (Stack ht. = 90 m)	Dutch National Model	Measured gas conc. (particulates) (ng/Nm^3)	43	216	406	331	NR	996	NR	140	236	432	272	NR	1080	2076
			Predicted max. annual ave. ground level conc. (pg/m^3)	0.129	0.648	1.218	0.993	NR	2.988	NR	0.420	0.708	1.296	0.816	NR	3.24	6.228

Page 1 of 2

TABLE 4.2.1.1L

ESTIMATED TOTAL ANNUAL OUTPUT OF PCDD AND PCDF FROM MUNICIPAL WASTE INCINERATORS (EXCLUDING ONTARIO)

Reference	Source Location/ Description	Matrix	ESTIMATED TOTAL ANNUAL OUTPUT PER INCINERATOR (g/yr)												
			PCDD						PCDF						Total PCDD + PCDF
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
Gizzi *et al.* (1982)	Italy(Como) modern incinerator with grate system for combustion and electrostatic precipitator (33,148 Nm^3/hr.)**	Total Stack emissions Ave.	37.8	57.8	106.4	83.1	36.6	320.8	89.8	72.6	91.2	62.5	36.0	352.2	673.8
		Low	2.0	3.8	5.5	3.8	5.5	20.6	8.7	7.8	6.4	8.4	5.5	36.9	57.5
		High	327.5	607.0	1105.6	838.0	183.4	3061.4	8269.8	657.0	850.8	410.9	111.0	2856.6	5918.1
Olie *et al.* (1982)	The Netherlands -average* of the 9 large (multiple oven) incinerators in the Netherlands (ave. 178,000 Nm^3/hr.)	Fly Ash	622	1689	4022	5067	2300	13700	1156	2078	3056	2089	344	8722	22,422
		Stack-collected particulates	89	378	689	544	700	2400	256	422	822	456	111	2067	4,467
SFOEP (1992)	Switzerland - Zurich Josefstrasse (modern plant with efficient furnace and abatement equipment). (94000m^3/hr)	Stack-collected particulates	2.8	8.2	18.7	18.2	39.1	87.0	12.8	16.4	11.8	8.0	6.3	55.3	142.3
		Stack-collected gaseous phase	0.5	0.9	1.8	1.6	1.3	6.2	5.6	6.1	3.6	2.2	0.4	18.0	24.1
		Total Stack Emissions	3.3	9.1	20.5	19.8	40.5	93.2	18.4	22.5	15.4	10.2	6.8	73.3	166.4
Tiernan *et al.* (1982a)	USA - Hempstead Resource Recovery Plant - N.Y. (not operating under ideal conditions.)	Total Stack Emissions	150	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR	NR

TABLE 4.2.1.1L

ESTIMATED TOTAL ANNUAL OUTPUT OF PCDD AND PCDF FROM MUNICIPAL WASTE INCINERATORS (EXCLUDING ONTARIO)

Reference	Source Location/ Description	Matrix	ESTIMATED TOTAL ANNUAL OUTPUT PER INCINERATOR (g/yr)												
			PCDD						PCDF						Total PCDD & PCDF
			T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total	
Olie *et al.* (1983)	The Netherlands - Zaanstad Incinerator (stack ht. = 90 m.)	Stack-collected particulates	34.6	173.7	326.5	266.2	NR	800.9	112.6	189.8	347.4	218.7	NR	868.4^a	1,669.3^a
Redford *et al.* (1983)	USA - Resource Recovery plant 17,000 kg raw refuse per hr. at 650°C	Total Stack Emissions	5	NR	10	6	2	23	70	NR	50	6	0.5	126.5	149.5

NR - not reported.

* - average results on a per incinerator basis derived by dividing total annual output for country by number of incinerators in the study (9).

** - flow rate calculated from data presented in paper and used to calculate total annual output.

a - totals exclude O_8CDD and O_8CDF

Summary

On the basis of the extensive literature dealing with the formation of PCDDs and PCDFs from the incineration of raw and treated municipal waste, it is apparent that the synthesis of these compounds is influenced primarily by fuel characteristics and process operating conditions. In this regard, poor combustion conditions and the presence of PCDD precursors or chlorine in the fuel can be expected to promote formation of PCDDs and PCDFs during combustion. In addition the following summary can be made with respect to the formation of PCDDs and PCDFs in the incineration of raw and treated municipal waste.

1. Most combustion theories implicate the formation of PCDDs and PCDFs either with the pyrolysis of chlorinated organic precursors already present in the refuse/sludge or to de novo synthesis involving chemically unrelated (non-chlorinated) organics and an inorganic chlorine source. The main support for the de novo formation hypothesis is that so far no strong correlations between chlorinated organic precursors in the fuel and PCDD/PCDF formation have not been found. There are no positive monitoring data which demonstrate de novo formation; however, laboratory experiments show such mechanisms are possible.

2. Regardless of the formation mechanism(s) chlorobenzenes and chlorophenols are suspected of being major intermediate compounds.

3. **On the basis of laboratory studies and considering the limitations involved in large scale combustion, it is generally apparent that proper operation of municipal incinerators within their design criteria and careful control or regulation of operating parameters represents the most attractive abatement strategy for reducing PCDD and PCDF emissions.**

4. **Although several attempts have been made to relate PCDD and PCDF emissions to various operating parameters or feed composition, the only correlation reported (Italian incinerator) has been higher emissions with minimal combustion temperature.**

5. **Ontario studies have shown that PCDDs/PCDFs are found in the feedstock to municipal incinerators. This should be an important consideration in future investigations to determine PCDD/PCDF mass balances or formation/destruction mechanisms.**

In an effort to synthesize the large number of reports which have been published on the release of PCDDs and PCDFs from the incineration of raw and treated municipal waste the global data base has been tabularized and assessed for physical partitioning and congener group distribution trends. It must be emphasized that this type of analysis is subject to a number of caveats, the primary ones being great differences in sampling techniques, stack gas temperature and the effect of various sampling artifacts on physical partitioning and isomer group distribution. As such the following summary statements should be considered as very approximate estimates of emission trends.

1. On a global basis (including Ontario) total PCDDs and PCDFs (T_4-O_8) appear to be equally distributed in precipitated fly ash averaging about 1 ug/g each; however from the Ontario data base total (T_4-O_8) PCDD concentrations in precipitated fly ash, are with one exception, about 15 x lower, averaging about 80 ng/g. In two of the three Ontario studies where both PCDDs and PCDFs in fly ash were analyzed, the total PCDF level ranged up to 2 x higher than the corresponding total PCDD concentration.

2. On a global basis about 50% of the total PCDD in precipitated fly ash consists of the H_6- and O_8CDD isomer groups, with the remaining 50% divided between the T_4-, P_5- and H_7CDD isomer groups. However, from the Ontario data base approximately 45% of the total PCDD in precipitated fly ash consists of the H_6 isomer group with the T_4-, P_5- and H_7CDD averaging about 15% each.

3. On a global basis over 90% of the total PCDF in precipitated fly ash consists of the T_4- to H_7- isomer groups (20-30% each) with a much lower (5%) proportion present as O_8CDF. In the Ontario studies, H_6CDF accounted for about 45% of total PCDFs with T_4-, P_5- and H_7CDF accounting for almost all of the remainder.

4. On a global basis about 3% of the total combined PCDD plus PCDF in precipitated fly ash is present as the T_4CDD isomer group.

5. On a weight:weight basis (ng/g) total PCDDs and PCDFs in stack-collected particulates (escaping the pollution abatement devices) are

on average, about twice and seven times as high as corresponding results for precipitated fly ash. These global results contrast with an earlier estimate of 10 x for PCDDs as reported in the NRCC (1981a) document and are even further contrasted with the ratio of over 500 x for a municipal refuse incinerator in Ontario.

6. On the basis of all studies in which total stack emissions (particulate plus gaseous or total sampling train) have been reported, the average PCDD and PCDF concentrations are both 4 ug/m^3. However, when only those studies which reported results for all five isomer groups of both PCDDs and PCDFs were considered, PCDDs comprised only 35% of total PCDDs plus PCDF emissions.

7. On a global basis, about 5% of the total PCDD plus PCDF in total stack emissions consists of T_4CDD, while T_4CDF comprises about 30% of total PCDD plus PCDF stack emissions. In the four Ontario studies in which total stack emissions have been presented as chlorinated isomer groups, the proportion of T_4CDD and T_4CDF in total PCDD plus PCDF stack emissions was similar to the global trend with average values of 8 and 26%, respectively.

8. In all studies (global data base) in which both particulate and gaseous results have been reported, total PCDD and PCDF gaseous concentrations comprised about 75% of the total stack emissions. In the 1984 Ontario study (SWARU, Hamilton) the gaseous and particulate partitioning was about 50:50.

9. **In isomer-specific studies, stack emissions and precipitated fly ash have been shown to contain about 30 PCDD and 40 PCDF congeners in the T_4-O_8 chlorinated isomer series.**

10. **The major T_4CDD isomers present in precipitated fly ash and stack emissions are 1,3,6,8- and 1,3,7,9-, present at about 20% and 12%, respectively of total T_4CDD; the most toxic isomer (2,3,7,8-T_4CDD) comprises on average, about 5% of total T_4CDD, 0.4% of total PCDDs (T_4-O_8) and 0.2% of total PCDDs plus PCDFs.**

4.2.1.2 Chemical Waste Incineration

In 1978 the first reports of PCDD emissions from the combustion of chemical wastes in Switzerland (Buser et al., 1978; Buser and Bosshardt, 1978) and the USA (Dow Chemical Co., 1978), were published. The results (Table 4.2.1.2A) which have been described in review articles (Esposito and Watkins, 1980; Little, 1981), confirmed that atmospheric contamination from these types of sources may be an important route of public exposure to PCDDs and PCDFs. The results of numerous micro-scale laboratory experiments involving the pyrolysis of chlorophenols, chlorobenzenes and PCBs (Buser and Rappe, 1978; Buser et al., 1978; Buser, 1979; Buser and Rappe, 1979; Buser and Bosshardt, 1978 and Mazer et al., 1983) also have confirmed the formation of PCDDs and PCDFs. It also was pointed out by Esposito and Watkins, 1980) that although unbound PCDDs are destroyed by incineration at a temperature of 800°C, T_4CDDs and other chlorinated congeners bound to particulate matter are essentially unaffected by incineration temperatures up to 1150°C and can escape unaltered.

Since 1978 there have been a number of extensive investigations into the release of PCDDs and PCDFs from the incineration of chemical wastes. The results of these studies are summarized in Table 4.2.1.2A.

The most comprehensive study involved two different rotary kiln facilities located in Texas and Arkansas, USA (Tiernan et al., 1982b). The wastes incinerated included chlorinated hydrocarbons (vinyl chloride, still bottoms), liquid PCBs, PCB-contaminated capacitors and various unidentified pesticide, paint and ink manufacturing chemicals. In both cases the PCDD and PCDF content (ng/sample) in the emission samples analyzed generally was at or below the analytical detection limits for the various chlorinated isomer groups. An exception to this general finding was the fairly common presence of T_4CDDs and T_4CDFs at levels marginally in excess of the detection capabilities. In two cases where detectable T_4CDD levels were apparent, isomer specific analysis revealed that 1,3,6,8- T_4CDD was the most abundant isomer (20-25%) with lower levels of 1,3,7,9-, 1,3,6,9-, and 1,2,7,9-T_4CDD being detected. In no case, however, was any 2,3,7,8-T_4CDD confirmed. Although the results were not quantified in terms of stack gas volume or total annual output, the authors did indicate that the EPA had ruled that the risks of additional cancers in the population exposed to the emissions were negligible and therefore granted approval for the plants to continue operating. The authors further concluded that under proper conditions (presumably high combustion temperatures and sufficient residence times - Table 4.2.1.2A), controlled incineration is a viable procedure for the destruction of PCB wastes.

Page 1 of 4

TABLE 4.2.1.2A

PCDD AND PCDF CONCENTRATIONS IN COMBUSTION EFFLUENTS FROM INDUSTRIAL CHEMICAL WASTE INCINERATION

REFERENCE	MATERIAL COMBUSTED	SOURCE/ DESCRIPTION	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Buser & Bosshardt (1978)	Used industrial oils	Industrial Heating Facility - Switzerland	Fly ash (ng/g)	100	160	180	130	40					
Dow Chemical Co. (1978)	Tars & natural gas	Waste Incinerators - Midland, Mich., USA Stationary Tar Burner	Stack Collected Particulate-Ave (ng/g)	2,3,7,8- ND	Other Isomers ND	8	92	300					
	Tars, Solid Waste & Natural Gas	Rotary Kiln (without supplementary fuel)	Stack Collected Particulates Ave. (ng/g)	2,800	5,500	21,000	16,000	263,000					
	Tars, solid waste, natural gas	Rotary Kiln (with supplemental fuel)	Stack Collected Particulate Ave. (ng/g)	2,3,7,8- ND	Other Isomers ND	2.0	32.0	230.0					
Hryhorczuk (1981)	Electrical wires, cable, x-ray films	Wire Reclamation Incinerator - Illinois, USA	Ash from base of stack (ng/g)	58					730				
			Ash from top of stack (ng/g)	410					11,600				

TABLE 4.2.1.2A

PCDD AND PCDF CONCENTRATIONS IN COMBUSTION EFFLUENTS FROM INDUSTRIAL CHEMICAL WASTE INCINERATION

REFERENCE	MATERIAL COMBUSTED	SOURCE/ DESCRIPTION	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Tiernan *et al.* (1982b)	Test 1 Liquid chlorinated hydrocarbon wastes (vinylchloride, still bottoms)	Rollins Environmental Services - Deer Park Texas, USA rotary kiln and liquid injection burner feeding common afterburner.	Total sampling train (ng/ sample)	6.94					13.5 (1.5)**				
	Test 2 Per Test 1, plus liquid PCB waste	Per Test 1	Total of all parts of sampling train (ng/sample)	1.42					22.0 (2.45)**				
	Test 3 Liquid PCB wastes and clean fuel oil	Per Test 1	Total sampling train (ng/ sample)	0					2.0 (0.2)**				
	Test 1 shredded capacitors (solid - no PCB) plus chlorinated hydrocarbons (pesticide process and paint/ink manufacturing waste)	Energy Systems Co. - El Dorado, Arkansas, USA rotary kiln and afterburner with 2 combustion chambers -	Total sampling train (ng/ sample)	0.263					2.0 (0.5)**				

* - 2,3,7,8-T_4CDD

** - 2,3,7,8-T_4CDF

TABLE 4.2.1.2A

Page 3 of 4

PCDD AND PCDF CONCENTRATIONS IN COMBUSTION EFFLUENTS FROM INDUSTRIAL CHEMICAL WASTE INCINERATION

REFERENCE	MATERIAL COMBUSTED	SOURCE/ DESCRIPTION	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Tiernan *et al.* (1982b) (Cont'd)	Test 2 Per Test 1, plus liquid PCBs and shredded PCB containing capacitors	Per Test 1	Total sampling train (ng/ sample)	0.476 0.476					6.0 (1.5)**				
	Test 3 Liquid PBC wastes and PCB containing capacitors mixed with diesel fuel	Per Test 1	Total sampling train (ng/ sample)	0					0 (0)**				
Chiu *et al.* (1983)	Aroclor 1254 (PCB) mixed with methanol	Experimental Plasma Reactor - Canada Feb., 1982	Stack gas (ng/dscm)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
Leung and Meadows (1984)	Askarel (20% by-weight Aroclor 1260)	June, 1983	Stack gas (ng/dscm)	ND	ND	ND	ND	ND	ND	2500	400	10	ND

* - 2,3,7,8-T_4CDD
** - 2,3,7,8-T_4CDF

TABLE 4.2.1.2A

PCDD AND PCDF CONCENTRATIONS IN COMBUSTION EFFLUENTS FROM INDUSTRIAL CHEMICAL WASTE INCINERATION

REFERENCE	MATERIAL COMBUSTED	SOURCE/ DESCRIPTION	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Reppe **et al.** (1983a)	pentachlorophenol	**Industrial Boiler - USA**	Baghouse ash (ng/g)	960 (<5)*	1,400	2,000	700	200	900 (100)**	1,500	150	60	6
			Bottom ash (ng/g)	10 (<1)*	20	40	100	140	ND (ND)**	ND	ND	ND	-
	PCB (Oil-fired)	Rotary cement kiln - **Slemmestad, Norway**	Electrostatic filter (ng/g)	<1.0	<1.5	<2.5	<0.3	<0.5	<1.0	<2.5	<2.5	<5.0	<0.5
			Stack Gas Condensate (ng/L)	<0.2	<0.1	<0.6	<0.2	<0.4	<0.2	<0.04	<2.4	<4.6	<1.0
			XAD Filter 1 (ng/g)	<0.1	<0.05	<0.7	<0.1	<0.2	<0.1	<0.05	<0.7	<1.4	<0.3
			XAD Filter 2 (ng/g)	<0.05	<0.05	<0.2	<0.05	<0.1	<0.2	<0.05	<0.3	<3.0	<0.3
			Filter 1 (ng)	<3.0	<6.3	<6.3	<0.3	<0.6	<6.3	<6.3	<6.3	<6.3	<1.3
			Filter 2 (ng)	<0.1	<0.2	<0.9	<0.1	<0.4	<0.1	<0.2	<0.9	<0.2	<0.4
			Sintered cement (ng/g)	<50.0	<50.0	<250.0	<5.0	<250.0	<50.0	<50.0	<250.0	<50.0	<250.0
Gross **et al.** (1983)	PCBs 20% chlorobenzenes, kerosene, pump oils, sulphur (250 ppm) (total 350 tonnes burned)	**M/T Vulcanus Incinerator Ship - Gulf of Mexico, USA**	Stack gas (excludes partic.) (ng/m^3)	ND					<0.3-<3.0				

* - 2,3,7,8-T_4CDD

** - 2,3,7,8-T_4CDF

The results of a study by Rappe et al. (1983a) in which PCBs were incorporated into an oil-fired rotary cement kiln in Norway, also support the use of high temperature incineration (1200-1300°C in solid material and up to 2000°C in flames) as an effective destruction technique for PCBs, as in no case (including the sintered cement) were any PCDDs or PCDFs detected the effluent (See Table 4.2.1.2A for detection limit).

Stack gas from an experimental plasma reactor in Canada also was found to be free of PCDDs and PCDFs (Table 4.2.1.2A) in an Aroclor 1254 (PCB) incineration project (Chiu et al., 1983). However, a second test of this facility did detect a total of 2900 ng PCDF/dscm in a short duration burn (Leung and Meadows, 1984).

Rappe et al. (1983a) did report finding levels of 5260 and 2616 ng PCDD and PCDF/g respectively, in fly ash from an industrial boiler in the U.S. in which pentachlorophenol had been incinerated; however, details of the combustion conditions were not published.

Other examples of chemical incineration include the reclamation of wire cables wrapped in oil-saturated paper (Hryhorczuk et al., 1981) and the incineration at sea of chemical wastes comprised of PCBs, chlorobenzenes and oils (Gross et al., 1983). In the case of the wire reclamation plant, T_4CDD and T_4CDF were definitely confirmed in the ash from the base of the stack with T_4CDF present in concentrations 12-28 times the corresponding T_4CDD concentrations (Table 4.2.1.2A). Although incineration temperatures were not recorded it was reported that the afterburner on the incinerator probably was not operating properly.

Chemical incineration at sea on the M/T Vulcanus appeared to represent a safe destruction mechanism for these materials (Gross et al., 1983,) as stack gas T_4CDD concentrations were below detection capabilities. T_4CDF values were reported to lie in the 0.3-3 ng/m^3 range with 2-7 isomers comprising these low levels. Incineration was conducted at a temperature of 1300°C with a 1.3 second residence time.

The importance of residence time in effective PCB destruction is underscored by the work of Ahling and Lindskog (1982) who found no relationship between temperature and PCB residues over a range of 675-1000°C, provided the transit time was sufficiently long.

Summary

On the basis of the foregoing investigations of actual industrial and experimental incinerator facilities and considering the micro-scale pyrolysis experiments, it can be concluded that:

1. **Under some conditions the potential exists for PCDD and PCDF formation and emission into the atmosphere during the incineration of waste chemicals.**

2. **With proper incinerator design and operation, chemical wastes can be effectively destroyed with negligible emissions of PCDD/PCDF to the atmosphere.**

Accordingly, each site (incinerator) must be examined on the merits of the operational parameters before any general PCDD/PCDF emission factors can be estimated.

4.2.1.3 Biological Waste Incineration

The major types of biological waste which are routinely disposed of by incineration include large amounts of pathological waste from hospitals and other medical or clinically oriented institutions, veterinary clinics and human/animal tissues at crematoria. Although no on-site incineration investigations have yet been conducted in Ontario, it is known, that the incineration of these types of wastes is a commonly accepted practice and that in many smaller institutions where efficient incineration facilities are not available, the waste is simply fed into the existing on-site central heating boiler. As the composition of these wastes would probably include chlorinated disinfectants, plastics and drugs, as well as the pathological waste, the potential for PCDD/PCDF formation exists. This has been confirmed in a recently reported study (Bumbaco, 1983) involving emissions from a hospital incinerator in British Columbia, Canada. The results are shown in Table 4.2.1.3A.

TABLE 4.2.1.3A

CONCENTRATIONS OF PCDDs AND PCDFs
IN STACK EMISSIONS FROM
ROYAL JUBILEE HOSPITAL INCINERATOR
VICTORIA, B.C. - 1983

Congener Group	Concentration (ng/m^3)	
	PCDD	PCDF
T_4	ND*	27.0
P_5	15.7	46.2
H_6	13.8	42.9
H_7	16.7	25.7
O_8	22.8	13.8
Total	68.9	155.6

* ND < 0.5 ng per sample

However, until such time as analytical investigations into PCDD/PCDF emissions are conducted in Ontario, any conclusion concerning the presence or magnitude of these compounds in stack emissions must remain tentative. The reduced use of hexachlorophene (a confirmed source of PCDDs) as a disinfectant no doubt has reduced the potential for PCDD/PCDF emissions.

4.2.1.4 Wood Combustion

Incineration of Chemically Treated Wood Products

The former commercial use of large quantities of chlorophenols in paper mills as wood preservatives (fungicides) and as slimicides and the previously established role of these chemicals in the pyrolytic formation of PCDDs and PCDFs has prompted a number of micro-scale laboratory investigations into the potential release of PCDDs and PCDFs from the combustion of chemically impregnated wood and wood products.

Probably one of the most important discoveries in the experimental wood pyrolysis studies was summarized by Ahling and Lindskog (1982). In open fire burning of wood it was found that chlorophenols (tetra and penta) and chlorobenzenes, were formed in fairly significant quantities from the combustion of this material. In an earlier detailed study (Jansson et al., 1978), several commercial and technical grade chlorophenate formulations were pyrolyzed with sawdust and wood chips at three different temperatures and under open fire conditions. The results (Table 4.2.1.4A) confirmed the formation of significant quantities of T_4-, P_5-

and H_6CDDs, depending on the chlorophenol formulation used. Increasing the temperature (from 500° to 800°C) and/or the residence time decreased PCDD emissions. Under open fire conditions total (T_4-O_8) PCDD concentrations in the combustion gases and particulates ranged from just over 30 ug/g for pentachlorophenol, to over 500 ug/g for formulations containing tetrachlorophenol. It also was concluded that the major T_4CDD isomer was 1,3,6,8- T_4CDD. On the basis of these findings the use of chlorophenol formulations is now prohibited in Sweden.

In another experimental investigation of wood shavings, Rappe et al., (1978) confirmed the formation of significant quantities of PCDDs in combustion gases and particulates following pyrolysis of a commercially available chlorophenol formulation with wood shavings (Table 4.2.1.4A). On an isomer specific basis this work confirmed the finding by Jansson et al. (1978) that the major T_4CDD isomers are 1,3,6,8- and 1,3,7,9-T_4CDD with only a minor peak being apparent for the more toxic 2,3,7,8-TCDD isomer. The major hexa isomer was 1,2,3,6,8,9-H_6CDD with lower levels of the toxic 1,2,3,6,7,8-and 1,2,3,7,8,9-H_6CDD and 1,2,3,7,8-P_5CDDs being present.

Another more recent micro-scale pyrolysis experiment (Tiernan et al., 1983b) involving the use of pine wood with a chlorine donor (HCl) confirmed the release of PCDDs and to a much greater degree (20X) PCDFs. Again the most prevalent T_4CDD isomers were 1,3,6,8- and 1,3,7,9-T_4CDD (Table 4.2.1.4A). The high levels of PCDFs (3 ug/g) appeared to be associated with the presence of a chlorine donor as the combustion of pine wood alone failed to yield any PCDFs at levels of detection in the ng/g range. These results contrast with those of Jansson et al. (1978) who reported finding significant quantities of PCDFs from the combustion of untreated wood.

In an effort to relate these laboratory findings to more realistic open air or incinerator conditions the findings of Olie et al. (1983) and Chiu et al. (1983) have been included in Table 4.2.1.4A.

In the case of the open air burning of P_5CP-treated wooden boxes (Chiu et al., 1983), total (T_4-O_8) PCDDs of 0.4 ug/g particulate matter were detected with 96% of this present as H_6 (29%), H_7 (21%) and O_8 (46%) isomer groups. The only PCDF congener detected on the particulate matter was O_8CDF at 0.025 ug/g. In his summation of these Ontario data, Williams (1982) converted the contaminant levels to ng/dscm as shown in Table 4.2.1.4A. Total particulate plus gaseous emissions from the open burning of the PCP treated boxes amounted to 138 and 8.2 ng/dscm for PCDD and PCDF, respectively.

In the other study (Olie et al., 1983), a pilot scale fluidized bed incinerator capable of burning 100 kg/hr, was utilized in the combustion of 60-year old painted wood, P_5CP-treated new wood and hypochlorite treated paper. The results of fly ash analysis (Table 4.2.1.4A) confirmed the presence of total (T_4-O_8) PCDDs at about 0.2 and 0.3 ug/g for the old painted and new, P_5CP-treated wood, respectively. A total of 0.2 ug/g PCDFs were detected in both cases.

TABLE 4.2.1.4A

Page 1 of 2

PCDD AND PCDF CONCENTRATIONS IN COMBUSTION EFFLUENTS FROM THE COMBUSTION OF CHEMICALLY TREATED WOOD/WOOD PRODUCTS

REFERENCE	MATERIAL COMBUSTED	COMBUSTION CONDITIONS	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Jansson *et al.* (1978)		Pilot Scale Incinerator											
	Commercial potassium 2,4,6-trichlorophenolate (36% 2,4,6-T_3CP, 40% T_4CP)	mixed with sawdust and burned with wood chips 580°C	Gas & Particulates (ug/g formulation used)	230	74	9	8	6					
		600°C		2.5	1.2	0.2	0.6	0.7					
		840°C		1.8	1.4	<0.4	ND	ND					
		OPEN FIRE		100	81	7	<1	<1					
	sodium 2,3,4,6-tetrachlorophenolate (6% 2,4,6-T_3CP, 31% T_4CP and 3% P_5CP)	(as above) 510°C	Gas and particulates (ug/g formulation used)	140	240	360	95	<2					
		690°C		10	43	53	10	ND					
		850°C		ND	14	17	ND	ND					
		OPEN FIRE		47	160	270	58	<5					
	pentachlorophenol 25% 2,3,4,6-T_4CP; 4.3% PCP	(as above) 515°C	Gas and particulates (ug/g of formulation used)	<0.2	<0.2	0.4	1.5	1.7					
		710°C		<0.2	<0.05	0.8	<0.2	<0.04					
		800°C		<0.2	0.5	0.6	<0.4	<0.08					
		OPEN FIRE		<0.2	0.9	2.0	9.0	19.0					

TABLE 4.2.1.4A

PCDD AND PCDF CONCENTRATIONS IN COMBUSTION EFFLUENTS FROM THE COMBUSTION OF CHEMICALLY TREATED WOOD/WOOD PRODUCTS

REFERENCE	MATERIAL COMBUSTED	COMBUSTION CONDITIONS	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Rappe et al. (1978) Cont'd	Severex on wood wool	as above	Smoke and gases (ug/g chlorophenate used)	96-210	120-357	110-347	29-65	1.2					
Chiu et al. (1983) Williams (1982)	PCP impregnated wooden boxes	Open Air Burning (Ave. of 5 tests)	Particulate (ug/g)	0.002	0.014	0.118	0.083	0.187	ND	ND	TR	TR	0.025
			ng/dscm	1.7	3.7	42	27	61			TR	TR	8.2
			Gaseous ng/dscm	ND	ND	ND	ND	3	ND	ND	ND	ND	TR
Olie et al. (1983)	60 year old painted wood	Pilot scale Incinerator (Fluidized bed oven)	Fly ash (ug/g)	0.029	0.044	0.055	0.020	0.029	0.105	0.066	0.026	0.012	0.008
	PCP treated new wood		Fly ash (ug/g)	0.047	0.061	0.115	0.043	0.058	0.097	0.072	0.042	0.019	0.011
	Hypochlorite treated paper		Fly ash (ug/g)	0.001	0.003	0.009	0.007	0.004	0.004	0.002	0.003	0.002	0.001
Tiernan et al. (1983b)	wood (pine) plus HCl vapor	Micro-scale laboratory pyrolysis (with air)	Combustion products (ug/g of wood burned)	0.020	0.005	0.010	0.070	0.050	0.140	0.055	0.220	1.20	0.40

Summary

On the basis of the foregoing laboratory and pilot-scale pyrolysis investigations, it is clear that the burning of wood and wood products contaminated with chlorophenols represents a significant potential source of PCDDs and PCDFs into the atmosphere. This situation is complicated by the fact that:

1. **Much of the burning of this type of material is conducted in 'open fire' situations where control of temperature and residence time is not possible, and,**

2. **The amount of chlorophenol treated wood/wood products burned in Ontario each year is unknown but is currently under investigation by Environment Canada.**

Combustion of Untreated Wood

The growing use of wood as a sole or ancillary residential heating fuel has resulted in a number of recent studies to assess the complex array of organic materials (Hall and DeAngelis, 1980) emitted during the combustion process and to determine their environmental impact (Cooper, 1980). A number of recent publications addressing the potential for PCDD and PCDF formation and emission from wood burning have been summarized in Table 4.2.1.4B.

In the most comprehensive investigation, the combustion of apparently uncontaminated residential wood supplies in several areas of the U.S.

(Nestrick and Lamparski, 1982) has resulted in easily measurable amounts of PCDD and PCDF components in the deposited soot and particulates from the combustion units. The average values (all regions) in this study for chimney particulates were 1.2, 6.9, 6.0 and 6.0 ng/g for the T_4-, H_6-, H_7- and O_8-CDDs, respectively. The percent concentration of 2,3,7,8-T_4CDD relative to total T_4CDDs ranged from 0 to 16% with an average of 5.0%.

Although this work corroborated the earlier, more limited study by Dow (1978) involving soot scrapings from fireplaces, the results do not agree with the work of Rudling et al. (1980) or Tiernan et al. (1983b), who were unable to detect PCDDs or PCDFs in the combustion effluents from either fireplaces, residential wood burners, a wood-fired heating facility, or under controlled laboratory pyrolytic conditions. Unfortunately, the limits of detection in these studies either were not reported (Rudling et al., 1980) or were fairly vague 0.03-0.5 ng/g (Tiernan et al., 1983b) thereby rendering a conclusive comparison of the results of limited value. More recent Canadian work on this matter (Wang et al., 1983) has confirmed the presence of measurable quantities of only O_8CDD in the flue gas particulates from a residential wood stove. These results have not yet been quantified.

Samples of bottom and chimney ash from two wood burning stoves, one open fireplace and from out-of-doors open-air burning were analyzed by the Ontario Ministry of the Environment (Clement et al., 1984). Although only untreated wood was burned, PCDD/PCDF were detected in all samples.

Large differences in total PCDD/PCDF and relative congener amounts were observed between samples.

Estimations of ambient air quality in the vicinity of wood-burning stoves has been attempted (Health and Welfare Canada and Environment Canada, 1983). However, these estimates must be considered very hypothetical because they are based on stack soot analysis and not on emissions.

Summary

Many questions still remain unanswered in regard to the formation and release of PCDDs and PCDFs into the atmosphere (wood type, furnace construction, operator bias). Additional quantitative information is required in order to establish a reliable estimate of the magnitude of the emissions.

It can, therefore, be concluded only that the combustion of wood for heating or aesthetic purposes does represent a potential source for the release of PCDDs and PCDFs into the atmosphere.

Additional research including analysis of emissions from wood burning devices as well as suspended particulates in the atmosphere during winter months is required to quantify this component of the total atmospheric loading of PCDDs and PCDFs.

TABLE 4.2.1.4B

PCDD AND PCDF CONCENTRATIONS IN EFFLUENTS FROM THE COMBUSTION OF UNTREATED WOOD

REFERENCE	MATERIAL COMBUSTED	COMBUSTION CONDITIONS	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
DOW Chemical Co. (1978)	Birch, oak, willow, ash and apple, plus some paper	Fireplace - A	Soot scrapings (ng/g)	0.37 (0.1)*		3.4	16	25					
	Hardwood - mostly oak	Fireplace B	Soot scrapings (ng/g)	ND(<0.04)(ND)*		0.23	0.67	0.89					
Rudling *et al.* (1980)	60:40 deciduous: coniferous wood	35 Kw central heating furnace with a pre-chamber	Gases and part-iculates (ng/g)	(ND)*					(ND)**				
	as above	as above with pre-chamber disconnected	Gases and part-iculates (ng/g)	(ND)*					(ND)**				
	50:50 deciduous: coniferous wood	Fireplace stove with glass shutters	Gases and part-iculates (ng/g)	(ND)*					(ND)**				
	as above	air tight stove with fan and heat exchanger	Gases and part-iculates (ng/g)	(ND)*					(ND)**				
Nestrick & Lamparski (1982)	Red oak	Residential Wood Stoves USA Eastern Region - USA	Partic. from (Low)	0.002(<0.0006)*		0.021	0.063	0.095					
			chimney (High)	7.8 (0.16)*		5.6	5.4	7.2					
			(ng/g) (Ave.)	2.0 (0.066)*		2.2	2.1	2.0					
	fir, oak	Western Region - USA	Partic. from (Low)	0.093 (0.006)*		1.3	0.96	0.49					
			chimney (High)	0.514 (0.02)*		9.2	5.8	5.2					
			(ng/g) (Ave.)	0.269 (0.013)*		4.3	3.2	2.2					
	ash, birch, oak	Central Region - USA	Partic. from (Low)	<0.0001(0.0008)*		0.003	0.013	0.024					
			chimney (High)	3.8 (0.2)*		65.3	39.0	37.0					
			(ng/g) (Ave.)	1.5 (0.006)*		14.2	12.6	13.8					

( )* - 2,3,7,8-T_4CDD specific analysis.

( )** - 2,3,7,8-T_4CDF specific analysis.

Page 2 of 2

TABLE 4.2.1.4B

PCDD AND PCDF CONCENTRATIONS IN EFFLUENTS FROM THE COMBUSTION OF UNTREATED WOOD

REFERENCE	MATERIAL COMBUSTED	COMBUSTION CONDITIONS	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Clement *et al.* (1984)	Oak and paper	Outside air burning	Bottom ash	0.8	4.2	7.2	11	10	2.2	7.6	8.2	11	1.7
		Wood-burning stove	Chimney ash above damper	ND	ND	ND	0.1	0.2	ND	ND	ND	ND	ND
			Bottom ash ( ) same location one year later	ND(0.1)	ND(3.0)	ND(10)	0.3(1.2)	2.6(0.9)	9.1(0.4)	2.2(4.6)	1.0(9.3)	0.7(1.0)	ND(0.1)
	75% - 80% Poplar	Wood-burning stove	Bottom ash	0.1	0.2	0.7	0.5	0.1	0.1	0.2	0.5	0.3	ND
	Unknown	Open Fireplace	Chimney ash above damper	ND	0.5	1.7	0.5	0.4	0.3	1.4	1.7	0.4	0.1
			Bottom ash	ND	ND	0.3	2.0	3.1	ND	ND	0.1	0.4	0.1

4.2.1.5 Fossil Fuel Combustion for Power Generation

Interest in fossil fuel combustion sources was initially generated as a result of the report by Dow (1978) which revealed that fly ash from a coal/oil-fired powerhouse at their Midland plant was contaminated with 38 ng/g TCDD of which less than 20 ng/g (detection limits) was the toxic 2,3,7,8-T_4CDD isomer. Concentrations of 2, 4 and 24 ng/g were reported for the H_6-, H_7- and O_8CDD isomer groups, respectively. Since then a number of other investigators (DeRoos and Bjorseth, 1979; Kimble and Gross, 1980; Harless and Lewis, 1982; Haile et al., 1982; Nestrick and Lamparski, 1982, and Chiu et al., 1983) have analyzed fly ash and, in one case, flue gas for PCDDs and PCDFs with the results in most cases revealing non-detectable (low pg/g) levels of the T_4-O_8 PCDD and PCDF isomer groups. These limited data on the emission of PCDDs and PCDFs from fossil fuels (coal and oil) are presented in Table 4.2.1.5A.

In the most recent of these studies, Chiu et al. (1983) have examined the PCDD and PCDF concentration of fly ash from both a large, modern coal-fired powerplant (2-150000 Kw units) and a smaller coal-fired central heating facility, both located in Canada. The results (Table 4.2.1.5A) confirmed the presence of measurable (1-32 ng/g) PCDDs and PCDFs in at least one fly ash sample and traces in others from the powerplant, while non-detectable levels were found in fly ash from the central heating facility. On the basis of these findings the authors concluded that the investigation of power plants as potential sources of PCDDs and PCDFs should continue with emphasis on

TABLE 4.2.1.5A

PCDD AND PCDF CONCENTRATIONS IN EFFLUENTS FROM FOSSIL FUEL COMBUSTION

REFERENCE	MATERIAL COMBUSTED	COMBUSTION CONDITIONS	SAMPLE MATRIX (UNITS)	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
DOW Chemical Co. (1978)	coal and fuel oil	Industrial power plant	fly ash (pg/g)	38,000		2,000	4,000	24,000					
DeRoos & Bjorseth (1979)	coal	power plant	fly ash (pg/g)	ND(<1.5)									
Kimble & Gross (1980)	low sulphur, high ash coal	large modern power plant	fly ash (pg/g)	0.6									
Mahle & Whitting (1980)	bituminous coal+ air	Micro-scale laboratory	(pg/g coal)	ND(<80)		ND(2100)	570	1,300					
Haile et al. (1982)	pulverized coal	4 coal-fired utility power plants	fly ash (pg/g) flue gas(pg/m^3)	ND(<10) ND(<100)	ND(50) ND(500)	ND(<50) ND(<500)	ND(<70) ND(<700)	ND(<10) ND(<700)	ND(<10) ND(<100)	ND(<50) ND(<500)	ND(<50) ND(<500)	ND(<70) ND(<700)	ND(<70) ND(<700)
Harless & Lewis (1982)	coal	7 coal-fired power plants	fly ash (pg/g)	ND(<1.0-2.8)									
Nestrick & Lamparski (1982)	oil	residential oil furnace	flue pipe soot (pg/g)	ND(<2.4)		ND(<4.2)	25	130					
Chiu, et al. (1983)	coal	large modern coal-fired power plant	fly ash (pg/g)	1,000	6,000	6,000	5,000	ND	8,000	32,000	18,000	18,000	ND
		coal-fired central heating facility	fly ash (pg/g)	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

determining the relationship between the emission concentrations and variables such as chlorine content of coal and combustion conditions. In a more complete presentation of the power plant sampling results (APCD, 1981), total particulate phase stack emissions for the large power plant and a smaller central heating facility were reported (Table 4.2.1.5B).

TABLE 4.2.1.5B

TOTAL PARTICULATE - PHASE STACK EMISSION CONCENTRATIONS OF PCDDs AND PCDFs FROM TWO CANADIAN COAL-FIRED SOURCES*

SOURCE	PARTICULATE-BORNE STACK EMISSIONS (ng/dscm)											
	PCDD						PCDF					
	T_4	P_5	H_6	H_7	O_8	Total	T_4	P_5	H_6	H_7	O_8	Total
Large coal fired power plant	0.1	0.1	ND	0.1	ND	0.3	0	1	2	ND	ND	3
Smaller coal fired central heating facility	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND	ND

* APCD - 1981
(dscm = dry standard cubic metre)

On a mass-emission basis, total (T_4-O_8) PCDD and PCDF output to the environment for the large coal-fired power plant which burned about 726,000 tonnes of coal annually was 15 and 51 mg/day or 2 and 19 g/yr, respectively.

On the basis of the limited work (1 study) with oil-fired residential furnaces (Nestrick and Lamparski, 1982), it would appear that PCDDs (H_7 and O_8) are present in minimal quantities in combustion residue from these sources. As with coal, additional verification on a larger scale will be required to confirm these findings.

Summary

On the basis of the above information, it can be concluded that regarding power-generating facilities only coal-fired power plants have demonstrated any significant potential as a source of atmospheric PCDDs and PCDFs. In view of the limited data base investigation of coal and oil-fired power plants as sources of PCDDs and PCDFs should continue.

4.2.1.6 Other Combustion Sources

Other combustion sources which represent potential atmospheric sources of PCDDs and PCDFs have not been well studied or quantified but do warrant consideration based on their potential for either short-duration, high level or on-going, low level atmospheric input. The general information which has been published has been summarized briefly for each emission type.

Electrical Capacitor/Transformer Fires

The pyrolysis of commercial mixtures of PCBs, common components of electrical capacitors, has been shown under controlled combustion conditions to yield significant quantities of PCDFs in the range of 1-5% (Rappe et al., 1982), suggesting that uncontrolled burning of PCBs can represent a significant environmental source of PCDFs. This concern has been validated following studies of a number of electrical capacitor fires which involved the post-fire analysis of the combustion products via wipe tests of exposed surfaces.

In an investigation following a transformer fire in Binghamton, New York, involving dielectric fluids consisting of 65% PCB and 35% chlorinated benzenes, soot was found to be contaminated with a total (T_4-O_8) PCDF concentration of 2160 ug/g. The most toxic isomers 2,3,7,8-T_4CDF, 1,2,3,7,8- and 2,3,4,7,8-P_5CDF and 1,2,3,4,7,8- and 1,2,3,6,7,8-H_6CDF were found to be the major constituents within each chlorinated isomer group. Unlike most other electrical capacitor/transformer fires which involved the combustion of mineral oils and PCBs, the Binghamton fire also involved chlorinated benzenes and as a result a total of 20 ng/g PCDDs were detected in a soot sample (Rappe et al., 1983a).

Some data exist for levels of PCDF in soot from a transformer fire in Toronto involving Aroclor 1260. The study (Quilliam, 1978) analyzed soot resulting from the combustion of PCBs and compared the findings to levels of PCDF in the original transformer oil and to pure Aroclor 1260. The data are summarized in Table 4.2.1.6A.

The soot sample contained a total average concentration of 5 ug/g PCDF (0.7-10 ug/g). Based on the amount of PCB in the samples (1% in the wet soot; 50% in the transformer oil), this represents a 6000-fold increase in concentration. This indicates that oxygenated products were formed during the fire and did not originate directly from the transformer oil. There was no evidence of PCDD formation.

On the basis of these results it can be concluded that accidental combustion of transformer oils containing Aroclors can be considered a direct input of PCDF into the environment. Although atmospheric monitoring has never been reported during an accidental fire/explosion of an electrical capacitor or transformer, the results of wipe tests following these incidents suggest a fairly limited dispersion of the contaminants (Jansson and Sundstrom, 1982).

TABLE 4.2.1.6A

ESTIMATES OF CHLORINATED DIBENZOFURANS FROM TORONTO TRANSFORMER FIRE[a]

Tentative Identity	Estimated Concentrations (ug/g) Wet Soot Sample	Transformer Oil	Arochlor 1260 Std.
D_2CDF	0.02-0.2	N.D.	N.D.
T_3CDF	0.2 -1.6	N.D.	N.D.
T_4CDF	0.4 -3.7	N.D.	N.D.
P_5CDF	0.1 -4.7	N.D.	N.D.
H_6CDF	0.03-0.3	N.D.	N.D.
H_7CDF	0.01-0.05	N.D.	N.D.
PCB Concentration	1%	50%	100%
PCDF per PCB present (ug/g)	100-1300 (median = 600)	0.02-0.2 (median = 0.1)	N.D.

N.D. = not detected (0.01 ug/g = detection limit).

[a] source - Quilliam (1978).

In view of the frequency of PCB-transformer fires in the U.S. — at the Binghamton State Office Building, New York (February, 1981); One Market Plaza, San Francisco (May, 1983); and the First National Bank, Chicago (September, 1983) — and considering the element of risk of exposure of humans to PCBs and certain oxidation products, including PCDFs, the U.S. E.P.A. issued an Advance Notice of Proposed Rulemaking (ANPR) related to further control measures on the use of PCB-containing equipment (49 Federal Register 11070, March 23, 1984).

Forest Fires

As a result of the unpredictable nature of forest fires and the inherent danger involved in obtaining representative smoke samples, very few studies have been conducted to quantify the nature of the organic emissions. In the case of PCDDs and PCDFs there is a complete absence of air monitoring data. The approach which has been used to a limited extent following the controversy over the use of 2,4,5-T herbicides in the 1960s and 1970s, has involved the experimental combustion of herbicide-treated wood and vegetation and the analysis of the combustion effluents. In a recent review paper, Norris (1981) concludes that the amount of T_4CDD produced by the combustion of sprayed vegetation depends to a major degree on when the burning occurs after treatment. In this regard, burning which took place 1 and 3 months after herbicide application resulted in T_4CDD levels of 14 and 0.2 pg/g, respectively. In another study (Ahling et al., 1977) it was estimated that the formation of T_4CDD during a forest fire directly after 2,4,5-T herbicide application could be about 1 ug/m^2.

The work on the combustion of untreated wood (Section 4.2.1.4) also suggests that natural fires could represent potential sources of PCDDs and

PCDFs. However, due to the extreme variability in combustion conditions during a fire of this size, positive confirmation of forest fires as an atmospheric source must await the analysis of atmospheric smoke emissions during actual fires.

Automotive Exhaust and Cigarette Smoke

Although both of these categories were identified as potential sources of T_4CDD in the 1978 report by Dow and were referenced subsequently in the 1981 NRCC document, there has been no further study conducted either to support or negate the Dow results. These activities must, therefore, remain as potential but unconfirmed sources of PCDDs and PCDFs pending further research in this area.

4.2.1.7 Estimates of Atmospheric Input to the Ontario Environment from Combustion Sources

Municipal Incinerators

In 1981 the NRCC report estimated the total annual PCDD input to the atmosphere of Ontario from municipal incinerators at 2.2 kg. This was based on the PCDD concentration in precipitated fly ash as determined by Eiceman et al. (1981), the estimated particulate emissions from municipal incineration in the province as published by the A.P.C.D. -Environment Canada (1976) and a multiplication factor of 10 times to correct for the PCDD concentration in emitted particulates compared to precipitated fly ash (Lustenhouwer et al. 1980). No attempt was made to estimate non-particulate, gaseous emissions, nor were any studies conducted to verify the estimated (factored) emission figures.

On the basis of the fairly extensive monitoring which has been conducted throughout the world, it would appear possible to now amend this value and to include a factor for gaseous emissions. As indicated in Section 4.2.1.1 the global data suggest that a more appropriate conversion from precipitated fly ash concentration to corresponding values for emitted particulate matter is a factor of 2. However, on the basis of the Ontario study (Ozvacic et al. 1984b), the use of 2 or 10 as conversion factors both would significantly underestimate the conversion of precipitated fly ash concentrations to emitted particulate matter concentrations as the ratio in the Ontario study was about 500:1. Furthermore, the task of calculating the gaseous component also is complicated by the fact that the gaseous:particulate ratio in the Ontario study differed from the global ratio (50:50 for Ontario; 75:25 on global basis).

For these reasons and considering the fact that previous Ontario incinerator particulate emission data are 8 years out of date (APCD-Environment Canada, 1976) it has been concluded that the emission factor conversion method cannot be considered a viable technique for calculating annual atmospheric PCDD and PCDF outputs from the incineration of municipal waste in Ontario.

Since the publication of the 1981 NRCC report (1981a), a number of stack emission tests for PCDD and PCDF have been conducted on two main municipal refuse incinerators (SWARU, Hamilton and Commissioner Street, Toronto) and one (Ashbridges Bay, Toronto) of several sewage sludge incinerators in Ontario. The data from these tests together with the emission parameters documented in the various test runs are shown in Table 4.2.1.7A.

On the basis of these parameters and the daily operational status of the respective incinerators, an estimate of the total annual output of PCDD and PCDF from these three facilities via stack emissions has been calculated. As is apparent from Table 4.2.1.7A, these numbers are clearly in excess of the NRCC estimate of 2.2 kilograms of PCDD per year.

The difference between the two methods is even greater considering the fact that in the case of the SWARU data, the emission estimates in the foregoing table are based on the operation and testing of only one of the two boilers and therefore should be doubled to reflect the total contribution to the environment from this source. The estimated output from the Ashbridges Bay sludge incinerator also is believed to underestimate actual output as outlined in the note on Table 4.2.1.7A. Making the two-fold adjustment for the SWARU emissions and using the average of the two separate testing programs at SWARU, the total (T_4-O_8) PCDD and PCDF emissions for the three Ontario sources amount to approximately 8 and 14 kilograms per year, respectively, for a total PCDD plus PCDF emission of 22 kilograms per year. In the case of PCDDs, this level (based on three incinerators) is approximately four times higher than the NRCC (1981a) estimate for all municipal waste incineration sources in Ontario.

Since the monitoring data are confined to these three Ontario sources, an estimate of total PCDD/PCDF emission for all Ontario waste cannot be

TABLE 4.2.1.7A

CALCULATED ANNUAL EMISSIONS OF PCDD AND PCDF FROM TWO MUNICIPAL WASTE AND ONE SEWAGE SLUDGE INCINERATOR IN ONTARIO

Incinerator (date monitored)	Stack Gas Conc. (ug/dscm) PCDD	Stack Gas Conc. (ug/dscm) PCDF	Stack Flow Rate (dscm/sec)	Oper'tl Status	Calculated Annual Emissions (g/yr.) PCDD	Calculated Annual Emissions (g/yr.) PCDF	Calculated Annual Emissions (g/yr.) PCDD + PCDF
Commissioner St.[a] Toronto							
Dec. 1981	0.3	0.4	82.8	24 hrs./day 7 day/week	682	911	1593
Dec. 1981	1.2	1.3	80.5		2914	3337	6251
Dec. 1983	1.6	1.4	86.6		4488	3870	8358
				(Ave.)	2695	2706	5401
Ashbridges Bay[b] Toronto							
Apr. 1982	0.5	0.9	31.7	24 hr./day 7 day/week	482	888	1370
Apr. 1982	1.1	0.5	31.7		1137	500	1637
Apr. 1982	0.6	2.2	31.5		589	2227	2816
				(Ave.)	736	1205	1941
SWARU[c] Hamilton							
May 1982	8.6	16.3	18.3	24 hr./day 5 day/week	3555	6693	10248
May 1982	9.7	35.6	17.2		3736	13744	17480
June 1982	5.8	9.1	19.0		2467	3899	6366
				(Ave.)	3253	8112	11365
SWARU[c] Hamilton							
May 1983 (Ave. of 13 tests) (Ave.)	2.7	6.2	16.8	24 hr./day 5 day/week	1015	2321	3336
(High)	7.2	10.3					
(Low)	1.1	3.0					

a - annual emissions may overestimate actual output as 3 units were operated during tests; normally only 2 are operated.

b - annual emissions underestimate actual output as tests were based on a common stack for only 2 units, other unit is older and emits from separate stack when in operation.

c - annual emissions underestimate actual output by a factor of about 2 as tests in 1982 and 1983 reflect monitoring in only 1 of the 2 separate stack flues (boiler #2 in 1982 and boiler #1 in 1983).

calculated. However, emissions from other sewage sludge incinerators operating in Ontario are considered minor since the Ashbridges Bay plant is the largest of its kind in Canada, total PCDD and PCDF emissions are estimated to be in the range of 10-12 and 18- 20 kilograms per year, respectively. The contribution of other incinerators located in apartment buildings (about 1,200 apartment building incinerators in Metropolitan Toronto alone, burning an estimated 40 million kg refuse/year) and in hospitals, department stores, etc. remains unknown, but certainly warrants additional research attention. Open fire combustion at some landfill sites also can be considered a potential source of atmospheric PCDDs and PCDFs.

Table 4.2.1.7B depicts measured stack emission data and the resulting estimated maximum half hour and annual average ground level ambient air concentration around incinerators for total PCDDs and PCDFs.

Measured PCDD and PCDF emission concentration distributions are listed in column 1 of Table 4.2.1.7B. Column 2 lists the calculated maximum ground level concentration (GLC) distributions. The GLC's are one-half hour average values. These GLC's are modeled estimates and are related to the emission numbers by a codified use of plume rise and diffusion equations (Regulation 308 R.R.O. 1980 under the Environmental Protection Act). The GLC's are a function not only of the emission numbers but also of stack height, temperature and diameter, as well as of atmospheric conditions. Column 3 lists the corresponding calculated maximum annual average concentration distributions. These are taken to be 1/15th of the 1/2 hour averages. The arbitrary constant of 15 was chosen conservatively

TABLE 4.2.1.7B

MAXIMUM HALF HOUR AND ANNUAL AVERAGE AMBIENT AIR PCDD AND PCDF CONCENTRATION NEAR THREE INCINERATORS IN ONTARIO

COLUMN 1	UNIT	COLUMN 2	UNIT	COLUMN 3	UNIT
Emission Concentration Distribution	ug/Nm^3[c]	Maximum ½ hr. Ground Level concentration Distribution	pg/m^3	Maximum Annual Average Ground Level Concentration Distribution	pg/m^3
U.C. Commissioner St(sampled Dec./81)and ASHBRIDGES BAY(sampled Apr./82) (Ozvacic, 1983)					
PCDD	0.7-1.0	PCDD	24-70	PCDD	1.6-4.7
PCDF	1.0-1.2	PCDF	39-71	PCDF	2.6-4.7
SWARU (sampled Apr./82) (Ozvacic, 1983)					
PCDD	8	PCDD	721	PCDD	27.4[a]
PCDF	20.3	PCDF	1826	PCDF	70[a]
SWARU (sampled May/83 with Controlled Operating conditions)(Ozvacic, 1984)[e]					
PCDD	2.6 ± 0.76[b]	PCDD	178	PCDD	9[d]
PCDF	6.1 ± 1.29[b]	PCDF	402	PCDF	19[d]

a Values obtained as shown: 1/2 hr. average x 1/15 x 4/7; taking into account not only the empirical factor of 15 but also the fact that incinerator operation took place 4 days out of every 7 days.

b Average parameters for thirteen tests, with both boilers operating.

c Nm^3 = normal cubic metre, i.e. at 0°C and 1 atmosphere pressure.

d Values obtained as shown: 1/2 hour average x 1/15 x 5/7; taking into account both the empirical factor of 15 and the fact that incinerator operation was 5 days of 7.

e Represent current operating conditions. Assumed to be worst case for Ontario incinerators.

from the range of 10 - 50. This range is an empirically determined set of parameters based on a comparison of 1/2 hour average and annual average values for other pollutants. It is a function of wind speed, wind direction, topographical features, thermal and mechanical turbulence and number of sources in the area.

On an isomer group output basis, all four Ontario studies have provided isomer group distribution of the emitted PCDDs and PCDFs (Table 4.2.1.1C). Table 4.2.1.7C depicts the calculated average annual PCDD and PCDF output (on an isomer group basis) for the Ontario incinerators.

TABLE 4.2.1.7C

CALCULATED AVERAGE ANNUAL OUTPUT* OF PCDD AND PCDF ISOMER GROUPS FOR MUNICIPAL INCINERATORS IN ONTARIO

	PCDD					PCDF				
	T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
	g/yr									
Commissioner St., Toronto	162	216	970	1078	270	562	441	909	612	179
Ashbridges Bay, Toronto	44	52	74	420	147	337	554	72	109	133
SWARU, Hamilton (1982)	1883	1027	182	130	26	5273	2758	42	16	16
SWARU, Hamilton (1983)	287**	268	261	113	86	996	854	399	79	23
Average (all tests)	594	391	372	435	132	1792	1152	355	204	88

* subject to caveats a) b) and c) as outlined in Table 4.2.1.7A.

** 2,3,7,8-T_4CDD comprises 6% of total T_4CDD, (Ozvacic et al., 1984b).

It is apparent from this Table that within the PCDDs and PCDFs the T_4 isomer group represents the greatest average output, with somewhat lower outputs being apparent for the P_5 - H_7 isomer groups for both PCDDs and PCDFs. In both cases the O_8 isomer group is lowest.

annual emission number due to daily and seasonal fluctuations in feed rates, feed composition, boiler operation as well as operational shut-downs. However, it is concluded that they do describe total annual emissions more accurately than the use of conversion factors as in the NRCC report (1981a).

Wood

Ontario's forest industry generates over 3 billion kg of sawdust, bark, wood chips and other types of mill residues per year (Ministry of Energy, 1983). As energy costs continue to escalate, the industry has intensified its energy-from-wood waste efforts, and now utilizes an estimated 2 billion kg in various combustion facilities. Wood also has increased in popularity as a residential heating fuel and it is now estimated that about 1.5 million full cords (5 billion kg) are burned annually in Ontario (White et al., 1981). The significance of the smaller residential combustion facilities is underscored by the fact that most of the particulate emissions from these sources are in the respirable (<2um) range (Cooper, 1980). In a recent study (Hall and De Angelis, 1980), the average particulate emissions for fireplaces and baffled/non-baffled stoves using seasoned and green wood types was determined to be approximately 3 g/kg wood burned. Applying this factor to the residential wood consumption figures for Ontario (5 billion kg) yields a total annual particulate emission of about 15 million kg.

Forest fires also must be considered as a massive contributor to total particulate emissions per year as over the past 10 year period (1972-1981) in Ontario an average of about 235,000 ha. have been burned per year. Estimates of annual particulates from wild and prescribed forest fires in Ontario are not available. However, on the basis of published information (Ward et al., 1976) for 4 geographical areas in the U.S. (Northeast, North Central, Rocky Mountain and Pacific) average total particulate emissions in Ontario probably are in the range of 400-500 million kg.

The combustion of chlorophenol-treated railway ties, shipping containers and other treated wood waste also represents an unknown but potential PCDD and PCDF input into the Ontario environment. This source is currently under investigation by Environment Canada.

Coal

Ontario is a major coal consumer accounting for about 18 million tonnes annually (Ministry of Energy, 1983). About 64% of this is used to generate electricity and 34% to produce steel. The remaining 3% is used for other industrial, residential and commercial purposes.

Although the emissions of PCDD/PCDF from coal burning facilities have not been studied in sufficient detail to permit the accurate estimation of total output to the environment, the results of one Canadian study (APCD, 1981) indicated an output of 2 and 19 g PCDD and PCDF (T_4 - O_8) per year, respectively from a power plant burning 726,000 tonnes of coal per year. Using these values, the total annual PCDD and PCDF output for all Ontario sources would be about 50 and 470 g, respectively. In view of these findings, the combustion of coal does not represent a major source of PCDD/PCDF into the environment. Even these fairly low numbers may overestimate actual output, as in several studies no PCDD or PCDF emissions were documented (Table 4.2.1.5A).

Pathological Waste

As indicated in Section 4.2.1.3, the combustion of pathological waste has not been confirmed as a source of PCDDs/PCDFs into the Ontario environment; however, considering the fact that these types of waste would contain chlorinated compounds, drugs as well as human and animal tissue, the potential for PCDD/PCDF formation under routine incineration conditions does exist. This is confirmed by the recent detection of PCDDs and PCDFs from a hospital incinerator in Victoria, B.C. (Bumbaco, 1983). In that test total PCDD plus PCDF emissions were reported to be 2650 ug per hour. At the feed rate during the four tests this amounts to a total PCDD plus PCDF emission of 2900 ng/kg refuse.

An estimate of the total amount of pathological waste incinerated in Ontario per year is not available; however, information from MOE-Regional offices suggests that a major urban hospital incinerates about 1-2 million kg per year of these and other refuse wastes. If the emission rate for the B.C. hospital is used this would equate to an output of 3 - 6 g of PCDDs and PCDFs in emissions per year for a major urban hospital.

Chemical Waste, Electrical Capacitors, Cigarettes, Automotive Emissions

No information which would assist in quantifying the possible emission of PCDD/PCDF from these source types in Ontario is available.

Summary

In view of the limited amount of information concerning the potential formation and release of PCDDs and PCDFs into the Ontario environment from the combustion of other materials including wood, coal, pathological waste, chemical waste, electrical capacitors, cigarettes and automotive fuels, it is not possible to derive any firm PCDD/PCDF output estimates. However, the information which is available concerning the combustion of these materials in Ontario has been presented in order to provide some relevance to their possible significance at a later date.

4.2.2 CHEMICAL MANUFACTURING AND USE

PCDDs and PCDFs have been shown to be present in at least trace amounts in a number of pesticide chemicals, pharmaceuticals and PCBs as a direct result of formulation from raw materials containing these impurities or their precursors. The specific reaction mechanisms for PCDD formation have been documented elsewhere (Esposito et al., 1980). Fishbein (1973) has summarized the types of PCDD produced during synthesis of some commercial pesticides. The present day levels of these contaminants are much lower (Section 4.2.2.2B).

Polychlorinated biphenyls (PCBs) have been widely used as dielectric fluids and heat transfer agents in electrical transformers and capacitors since 1930. PCBs also were used in a wide variety of common products such as hydraulic fluids, oils, paints, varnishes, resins, inks, carbonless paper, waxes, sealants, caulking compounds, plastics and adhesives (NRCC, 1978b). PCDFs have been identified in most PCB preparations (Table 4.2.2A).

TABLE 4.2.2.A

PCDF CONCENTRATIONS* IN AROCLOR, CLOPHEN AND PHENOCLOR
(Bowes *et al.*, 1975)

PCB	T_4CDF	P_5CDF	H_6CDF	TOTAL
Aroclor 1248 (1969)	0.5(25)	1.2(60)	0.3(15)	2.0
Aroclor 1254 (1969)	0.1(6)	0.2(12)	1.4(82)	1.7
Aroclor 1254 (1970)	0.2(13)	0.4(27)	0.9(60)	1.5
Aroclor 1260 (1969)	0.1(10)	0.4(40)	0.5(50)	1.0
Aroclor 1260 (lot. AK3)	0.2(25)	0.3(38)	0.3(38)	0.8
Aroclor 1016 (1972)	N.D.	N.D.	N.D.	1
Clophen A-60**	1.4(17)	5.0(59)	2.2(26)	8.4
Phenoclor DP-6**	0.7(5)	10.0(74)	2.9(21)	13.6

* - Expressed as ug/g PCB. Values in parentheses indicate percentage total dibenzofuran.
ND - not detected (<0.001 ug/g).
** - not used in North America

Input of PCDDs and PCDFs into the environment from chemical manufacture or use would be from:

i) Chlorophenols (mainly used in wood preservation),
ii) 2,4-D and related phenoxy herbicides,
iii) PCBs (mainly used in electrical equipment),
iv) Hexachlorophene (used as a hospital disinfectant).

4.2.2.1 Chlorophenol Production and Use

Uniroyal Chemical Division of Uniroyal at Clover Bar, Alberta was the last producer of chlorophenols in Canada ceasing production in July, 1983 (Jones, 1981; Environment Canada, personal communication). There are currently no Ontario manufacturers of chlorophenols; thus, no emissions containing PCDD from chlorophenol formulation are found in Ontario.

The chlorophenols imported and marketed in Canada are used either directly as pesticides, or as intermediates in the manufacture of pesticides

(Jones, 1981). For example, all of the 2,4-D_2CP produced in Canada was used as an intermediate in the manufacture of the herbicide 2,4-D. Annual Canadian usage of 2,4-D_2CP is about 4000 tonnes. 2,4-D_2CP is not produced in Ontario.

Although P_5CP and its sodium salt have a wide variety of applications as fungicides or biocides, which range from the preservation of wood to the treatment of water in cooling systems, the major use is as a wood preservative. This use accounts for 95% of the volume of P_5CP used in Canada (Jones, 1981).

Various estimates of the total annual Canadian usage of chlorophenols in wood preservation, eg., P_5CP, NaP_5CP, 2,3,4,6-T_4CP and Na 2,3,4,6-T_4CP, for the last decade have ranged from 2000 to 2500 tonnes (Jones, 1981; I.C.T.C., 1983). Ontario is estimated to use about one third of this amount (NRCC 18578, 1982).

In January, 1979, the Federal Government suspended the use of pentachlorophenol as a wood preservative and disinfectant in poultry houses. Also, limitations were imposed on the use of pentachlorophenol and sodium pentachlorophenate in leather tanning operations.

In 1981, Agriculture Canada announced the following suspensions in the use pattern of these chlorophenols:

a) as a wood preservative on interior woodwork of farm buildings, feed bins, troughs, silos and stalls;

b) as a wood preservative on all wooden food containers;

c) as a wood preservative in mushroom houses;
d) as wood preservatives and in stains for use in interior of homes;
e) as slimicides in pulp and paper operations;
f) for spray treatments for home and garden use;
g) as agricultural miticides and disinfectants;
h) for vegetation control as an industrial herbicide.

Pentachlorophenol is still registered as a wood preservative for exterior use, however, and can continue to be used for treating wood fences, poles, railway ties, etc. Manufacturers have been asked to change product labels, dropping the previously registered uses and adding the restriction "for exterior use only."

The U.S. E.P.A. issued its final regulatory position on wood preserving pesticides which includes pentachlorophenol and its salts on July 11, 1984. The following restrictions were placed on the application, use and PCDD content of pentachlorophenol (and its salts):

1) restricted to use by certified applicators or persons under their direct supervision,

2) label must state restricted use nature of the pesticide and include warning that exposure to women during pregnancy should be avoided,

3) Applicators must wear protective clothing. They may not eat, drink or use tobacco products during the application process. They must wash thoroughly after skin contact and before eating, drinking, smoking or using restrooms.

4) Closed systems for mixing and emptying powdered/granular formulations will be required within 3 years,

5) Pentachlorophenol may not be applied in homes or wood intended for use in interiors. The application of P_5CP to logs for construction of log homes is prohibited. P_5CP pressure-treated wood in a home is also prohibited.

6) If P_5CP is applied to wood where it will result in bare skin contact (e.g. on outdoor furniture), two coats of sealer such as urethane, shellac, latex epoxy enamel or varnish must be applied.

7) In farm buildings, P_5CP may be applied to support structures which are in contact with soil or other interior structures out of contact with farm animals if two coats of sealant are used.

   P_5CP should not be used where there may be contamination of feed, food, drinking or irrigation water.

   Shavings, sawdust, and treated wood should not be used for bedding, brooding facilities or food containers, etc.

8) Treated wood waste should not be burned in an open fireplace but may be disposed in ordinary garbage or buried. (Railroad ties may be burnt in industrial incinerators.)

9) Wood pressure-treaters will be required to send consumer information sheets to all places where treated wood is sold to instruct consumers about handling procedures and safe use of treated wood.

10) Registrants of P_5CP will be required to limit the H_6CDD content of P_5CP (and its salts) to 15 ppm (15 mg/kg) immediately and reduce that level to 1 ppm (mg/kg) or less within 18 months.

Manufacturers also must submit information on the manufacturing process including changes to lower H_6CDD, data on the analysis of product ingredients and information on the technical feasibility and costs of lowering H_6CDD below the 1 ppm (mg/kg) limit.

Manufacturers also must submit exposure data on home and farm spray applications of P_5CP to evaluate whether restricting use to certified applicators is sufficient to reduce risks.

USERS

In Ontario, there are two major users of pentachlorophenol in wood preservation:

i) Abitibi-Price, Northern Wood Preservers,	Thunder Bay
ii) Domtar Wood Preservers,	Trenton.

To a minor extent, Bay Wood Preservers, Thunder Bay, and Falconbridge Nickel Mines, Falconbridge, use P_5CP in wood preservation.

Analysis of technical P_5CP used in Canada in 1981 shows the presence of T_4CDD, H_7CDD and O_8CDD (Table 4.2.2.1A).

A Swedish study (Levin et al., 1976) showed that PCDFs concentrated in the sludge dipping tank by 3 to 10X.

TABLE 4.2.2.1.A

ESTIMATE OF T_4CDD - O_8CDD CONTAMINATION IN P_5CP USED IN ONTARIO WOOD PRESERVATION PLANTS

PCDD or PCDF Congener Group	Range of mg[1] PCDD or PCDF /kg P_5CP	Average mg[2] PCDD or PCDF /kg P_5CP	Annual[3] PCDD or PCDF Input (kg)
T_4CDD	<0.1	0.01	0.008
T_4CDF	<0.1	0.01	0.008
P_5CDD	<0.1	0.01	0.008
P_5CDF	<0.1	0.01	0.008
H_6CDD	1 - 4	2	1.6
H_6CDF	26 - 30	28	22.4
H_7CDD	53 - 125	85	68
H_7CDF	36 - 80	58	46.4
O_8CDD	190 - 650	420	336
O_8CDF	9 - 80	36	28.8 503 kg/year

1. Data of Jones (1981) and internal MOE data.
2. Midpoint of quoted range.
3. Based on 800 tonnes/year.

Similarly, a study by Lamberton et al. (1979) demonstrated that PCDDs accumulate in the recirculating P_5CP solution used in the Boulton process of wood treatment and in the sludge of the recirculating tank. The O_8CDD level was 34% higher in the recirculating P_5CP solution than in the original P_5CP. The O_8CDD content of the sludge was 90% higher relative to the fresh solution. It was not possible to determine whether the increase in PCDDs/PCDFs was due to 1) conversion of the appropriate precursors during the treatment process; 2) selective deposition of P_5CP in the wood leaving a dioxin enriched solution; 3) low solubility of the PCDD in the petroleum distillate used as the carrier; or 4) some other factor.

Environment Canada (private communication) has indicated that PCDDs and PCDFs are found in effluent samples and sludge samples from several unidentified wood preservation plants. Effluents from the plants were found to contain 21 to 424 ng PCDD/L and 2.6 to 99 ng PCDF/L. Sludge samples had a total of 2 to 11 ug PCDD/kg and 0.6 to 1200 ug PCDF/kg.

No Ontario data exist for levels of PCDDs or PCDFs in these sludges or recirculating liquids, or for effluents from wood preservation plants. It is, however, important to consider the PCDD and PCDF levels in these solutions in view of waste disposal from wood preservation plants (Section 4.2.3.2).

Pole Re-treatment

Treated wooden poles are widely used by Ontario Hydro and numerous other public utility companies in Ontario. These poles require treatment to prevent rotting especially below ground level. Such treatment involves hydraulically injecting about 4.5L of a grease, containing 10% P_5CP by weight, into the soil around the base of the pole every ten years or so. Ontario Hydro uses about 140,000 to 180,000 L of this 10% P_5CP grease annually. A recent study by the Nova Scotia Power Corp. gives details of the re-treatment procedure and the environmental dynamics of P_5CP in the soil surrounding the poles (NSPC, 1983).

Summary

Impact of PCDDs and PCDFs on the environment resulting from the use of chlorinated phenols will involve a) localized inputs to soil and water near wood preservation facilities and retreated utility poles, and b) more widespread soil and water inputs involving distribution and use of treated wood products. Direct inputs to the atmospheric environment are expected to be minimal.

4.2.2.2 Herbicide Production/Formulation/Use

2,4-D (2,4-dichlorophenoxyacetic acid) in its various formulations (amine or ester) is a selective herbicide employed for the control of broadleaved weeds. It is used mainly on grain crops (wheat, barley, oats, rye, corn) to control weeds such as wild mustard. It is also used for brush control on hydro power transmission line rights-of-way and to keep residential lawns and public parks free of weeds such as dandelions.

In Ontario, only Uniroyal Chemical, Elmira, produced 2,4-D as a variety of products including amine and ester formulations. Production was discontinued in 1970.

Cochrane et al. (1981) have documented the levels of D_2-, T_3-, and T_4CDD in the various formulations of 2,4-D used in Canada. (Table 4.2.2.2A).

TABLE 4.2.2.2A

LEVELS OF D_2-, T_3- and T_4CDDs IN 2,4-D FORMULATIONS IN CANADA (COCHRANE et al. 1981, NRCC 20647, 1983)

	PCDD ug/kg 2,4-D					
	D_2CDD		T_3CDD		T_4CDD	
Ingredients	Range	Average	Range	Average	Range	Average
Ester Products						
IOE(1)	100-8500(4)	1075	100-2500	1150	100-8700(4)	560
MBE (2)	156-24000(4)	1440	100-660	260	100-400	
Amine						
2,4-D Amine	ND(3)-409	45	ND-600	120	ND-300	45
Acid		ND		ND		ND

1) IOE = isooctyl ester
2) MBE = mixed butyl ester
3) ND = not detected; limit of detection (1 ug/kg)
4) Several high values were considered as outliers statistically and were not included in the calculation of average value.

Other phenoxy herbicides used in Ontario which may contain PCDDs are MCPA, MCPB and dicamba. Technical dicamba has been shown to contain about 20ug 2,7-D_2CDD/Kg. No 2,3,7,8-T_4CDD was detected at a detection limit of 2ug/kg technical dicamba (Forrette and Rozek, 1977). MCPA and MCPB may contain methylated PCDDs but no analyses of this class of PCDDs in herbicides have been reported. Estimates of the total amount of phenoxy herbicide used in Ontario range from 400 to 1200 tonnes per annum (Roller, 1979; OPAC, 1979; Krystynak, 1983). Of this total usage over 50% is 2,4-D, mainly the 2,4-D amine formulations.

In 1981, Agriculture Canada announced a maximum concentration of 0.01mg/kg for any dioxin congener in 2,4-D (Federal Trade Memorandum T-1-233). The federal Minister of Agriculture also pledged that only "dioxin-free" 2,4-D would be available by

1982. "Dioxin-free" in this context meant non-detectable at the 1ug/kg limit of detection.

Levels of PCDDs in 2,4-D technical products tested by Agriculture Canada in 1983 are shown in Table 4.2.2.2B. Current 2,4-D products contain 1 to 2 ug/kg 2,4-D or less of any PCDD congener (W.P. Cochrane, personal communication).

TABLE 4.2.2.2B

LEVELS OF D_2-, T_3- AND T_4CDDs IN 2,4-D FORMULATIONS IN CANADA IN 1983

Ingredient	PCDD (ug/kg 2,4-D)					
	2,7-D_2CDD		1,3,7-T_3CDD		1,3,6,8-/1,3,6,9-T_4CDD	
	Range	Average	Range	Average	Range	Average
2,4-D amine(41)*	ND-15	3.6	ND-16	2.4	ND-7	0.6
2,4-D ester (17)*	ND-11	3.8	ND-10	3.5	ND-6	2.0

* number of samples analyzed
ND = not detected; limit of detection (1ug/kg)

It will be noticed that PCDDs in phenoxy herbicides are mainly the lower chlorinated forms eg., D_2CDD to T_4CDDs. By comparison other sources of PCDDs such as incinerator emissions or chlorinated phenols contain more highly chlorinated congeners eg. T_4CDDs to O_8CDD. To estimate the amount of PCDDs entering the Ontario environment through use of phenoxy herbicides it is assumed that all mono- and dichloro phenoxy acetic acid-derived products eg. 2,4-D (ester or amine), 2,4-DB, 2,4-DP, MCPA, MCPB and dicamba contain 10ug PCDD/kg product or less. The annual estimate would then range from 4 to 12g total PCDDs.

Most of the phenoxy herbicides applied for weed or brush control are deposited on the target vegetation. Winddrift of aerial spray, and wind and water erosion can move some of the herbicide and presumably PCDD residues into the surrounding soil and/or water. Degradation of PCDDs in soil or on plants is described in Section 4.3.2.3.

Atmospheric input of PCDDs from phenoxy herbicide spraying operations could result in localized, short-term human and animal exposures. Phenoxy herbicide levels in air of 0.4 to 91.3ug/m^3 were measured in the spraying zone during Ontario forestry spraying operations (Libich *et al.*, 1984). Exposure decreases dramatically with distance from the spraying zone. Levels of exposure decreased by factors of 10^6 to 10^8 at 800m from the spray swathe (NRCC 20647, 1983).

Since current phenoxy herbicide formulations contain less than 10ug total PCDD/kg technical product, exposure of bystanders to PCDD from spraying operation is insignificant.

4.2.2.3 Previous Herbicide Manufacture and Use

Commercial use of 2,4-D on farms began in Canada in 1947. Statistics Canada indicate that for that year 346 tonnes were used. 2,4,5-T was introduced in 1950. The manufacture of 2,4-D and later 2,4,5-T products at Uniroyal Limited, Elmira from 1946 to 1969 is discussed in Section 4.2.3.1.

2,4,5-T produced prior to 1970 may have contained 16 to 27mg 2,3,7,8-T_4CDD/kg technical product. Up to this time 2,4,5-T had been registered for use around the home and on recreational areas. In 1970, Agriculture Canada canceled all registered uses of 2,4,5-T on lawns, recreational areas, and cereal crops. Precautionary labeling indicating a hazard to pregnant women was also required. 2,3,7,8-T_4CDD content of technical products was restricted to 0.1mg/kg 2,4,5,T at this time.

In the following decade, 2,4,5-T was mainly used for brush control on rights-of-way, on roadsides, on forests and for poison ivy control. In Ontario, government agencies used 50,180kg 2,4,5-T in 1977-1978 (OPAC, 1979).

In 1979 Agriculture Canada restricted the 2,3,7,8-T_4CDD content of 2,4,5-T products to less than 50ug/kg. In the same year the use of 2,4,5-T (and the associated 2,4,5-TP) in Ontario was restricted by requiring a special permit to use these products.

No permits were subsequently issued, although 2,4,5-T remained registered for use in Canada. Use of 2,4,5-T in Canada has declined substantially from over 50,000kg in 1973-1974 (NRCC, 1978a) to about 5,000kg in 1983 (I.C.T.C., 1983).

Consequently input of PCDDs to the Ontario environment from 2,4,5-T has been eliminated.

4.2.2.4 PCB Manufacture and Use

Monsanto, in the U.S., was the largest producer of Aroclors. PCDFs have been identified in all Aroclor preparations, except Aroclor 1016, as well as in Clophen A60 and Phenclor DP6 (Robinson and Smillie, 1977). Table 4.2.2.B summarizes the concentrations of T_4-, P_5- and H_6CDF in various

PCBs. No PCDDs have been detected in any PCB formulations; however, low levels of PCDDs may form following prolonged heating or high temperature episodes (section 4.2.1.6).

In 1971, Monsanto restricted sale of Aroclors to electrical equipment manufacturers. In 1976 production of Aroclors as transformer and dielectric fluids stopped. There are no known manufacturers of PCBs in Ontario. This product was always imported.

In 1977, Chlorobiphenyl Regulation No.1 (Federal Environmental Contaminants Act and Fisheries Act) prohibited new uses or installations of new electrical equipment containing PCBs. All surplus PCBs are now designated wastes.

Over 90% of all PCBs still in use in Canada are in Ontario. Current estimates of PCB quantities in Ontario are approximately 6.5 million litres or 9.75 million kg still in use in transformers and capacitors. Another 1.5 million litres or 2.25 million kg PCBs are in storage in 18 major centres and 160 smaller sites around the province awaiting disposal.

Proposals to manage PCBs as hazardous wastes are currently under review by MOE. Guidelines for the operation of mobile PCB destruction facilities are also being developed. Provisional ambient air guidelines of 450, 150 and 30 ug PCBs/m^3 for 1/2hr, 24 hr and annual average exposures have recently been developed. Guidelines for PCDD/PCDF emissions will also be applied to this activity.

Aroclors contain 1 to 2 ug total PCDFs/g PCB (Table 4.2.2.B); therefore, combined PCBs in use or storage around the province may contain 12 to 24 kg PCDFs. However, these PCDFs are in closed containers and do not represent a direct input to the environment.

Therefore, input of PCDF to the environment from use of Aroclor would be through:

i) old transformer leakage;
ii) fires involving transformers containing Aroclors;
iii) disposal in waste dumps of old transformer/ dielectric fluids containing Aroclors.

These amounts cannot be estimated from available information. No data for Ontario are available from which estimates of PCDF input to the environment from i) or iii) can be made. Section 4.2.1.6 discusses item ii).

4.2.2.5 Commercial/Domestic Products/Use

Under the Pest Control Products Act, P_5CP is available to the homeowner for the preservation of wood used for fences and external building construction. It is marketed as the main ingredient in wood preservatives for home use and also as an additive in stains and paints. Domestic sales are probably less than 1% of the overall PCP market (Jones, 1981).

Chlorophenols are also used in health care (dental) products and in disinfectants for home, farm and hospital use. Concentrations of P_5CP in three dental care products manufactured and used in Canada range from 0.1 to 0.22% (Jones, 1981).

Phenoxy herbicides are a common component of domestic weed control and lawn care products (OMAF Publication 529).

Hexachlorophene, derived from the condensation of 2,4,5- trichlorophenol, is also used in disinfectants for hospitals. It is incorporated into these disinfectants at concentrations of 0.006 - 3.0% (Jones, 1981). Hexachlorophene was found to contain 0.03 mg 2,3,7,8-T_4CDD or less/kg hexachlorophene. (Bowes et al., 1975). Recent information indicates that current hexachlorophene products may only contain the less toxic 1,2,3,4-T_4CDD isomer (A. Gilman - personal communication).

Summary

Input of PCDDs and PCDFs in domestic products such as paints, stains, wood preservatives, health care products, disinfectants, weed control and lawn care products cannot be estimated from the data reviewed. However, while the quantities involved may be low they will result in direct human exposure.

4.2.3 WASTE DISPOSAL SITES

The NRCC report (1981a) suggests that disposal sites where chemicals or chemical waste have been dumped can be considered as potential sources of PCDDs and PCDFs into the environment. Sites where soil or water contamination may occur are described in Sections 4.2.3.1 to 4.2.3.5. However, the magnitude of the atmospheric emission component is not discussed.

In their examination of the Hyde Park landfill site, New York, Esposito and Watkins (1980) point out that although T_4CDDs were detected in surface dust from the neighbouring industries they were not detected in air samples collected from either the plants or the landfill.

The aqueous migration of T_4CDD from the Hyde Park site has, however, been confirmed in environmental samples from a creek draining the area (Esposito and Watkins, 1980).

In a more recent report (Thibodeaux, 1983), the results of extensive investigations in the vicinity of a herbicide manufacturing facility (VERTAC) that practiced on-site disposal of 2,4-D and 2,4,5-T production wastes are summarized and used in the development of emission transport models. Although the models do validate the movement of T_4CDD via vaporization from contaminated soil and pond surfaces and entrainment of soil particles, it is difficult to separate these identifiable surface contamination results with the movement which is modeled on the basis of upward migration from landfill cells through an assumed 1 m thick soil cover and vaporization from the surface. As this type of migration/volatilization process would be required for the atmospheric escape of PCDDs and PCDFs from totally covered landfill sites, additional evidence in the form of on-site atmospheric monitoring is required to confirm that these chemicals are indeed escaping through landfill caps into the atmosphere.

Other factors which should be evaluated are the contribution of wind-blown dusts contaminated with PCDDs/PCDFs as wastes are trucked to landfill sites and dumped for subsequent burial, e.g., fly ash and bottom ash from municipal waste incinerators (Table4.2.1.1.D) and emissions from gas migration control systems in land fills which allow natural gases to escape. These gases are usually combusted in "flares".

4.2.3.1 Elmira

Background

Since 1941 the Uniroyal Chemical division of Uniroyal Ltd. in Elmira has produced a wide range of organic chemicals for the agricultural, rubber and plastics industries. From 1950 to 1969, Uniroyal produced the herbicide 2,4,5-T. The plant occupies a 40-acre site on the east side of the community, with Canagagigue Creek running north to south through the middle of the property. Uniroyal is Elmira's principal industry. (Facts, 1982).

Prior to 1964 Uniroyal wastes were blended in a series of lagoons located on both sides of Canagagigue Creek and neutralized before discharge to the creek. Sludges and concentrated wastes were burned (until 1970), stored on Uniroyal property or hauled to approved sites in Canada or the United States. After construction of the combined municipal-industrial sewage treatment plant in 1964 most, and eventually all, of Uniroyal's liquid wastes went to the treatment plant.

At the direction of the Ontario Water Resources Commission, operational lagoons and all temporary disposal pits were cleaned out. The lagoon wastes were buried in two plastic-lined pits on the east side of Canagagigue Creek. The operational lagoons on the west side of the creek were lined with clay bases and were returned to service, and a number of shallow test wells were established to monitor local groundwater quality.

Currently, processed liquid wastes receive equalization and pH control in aerated lagoons and then receive carbon treatment before discharge to the municipal sewage treatment plant. Concentrated liquid wastes and sludges are hauled to approved disposal sites. A new system (Wetox) is currently being installed to pretreat some of the more concentrated liquid waste streams. The product from the Wetox process will also receive carbon treatment before discharge to the municipal treatment plant. For the past 12 years, essentially no Uniroyal wastes have been added to on-site storage.

In addition to an extensive monitoring program being conducted on all of the municipal, domestic and Uniroyal test wells, two containment wells have been constructed on Uniroyal property, to control the groundwater flow in the deep aquifer. This system is designed to contain any contaminants and prevent off-site migration in the deep aquifer.

While there appeared to be no visible leakage from the pits and lagoons, samples from test wells installed by Uniroyal have shown the presence of chlorophenolic compounds in the shallow and deep groundwater.

Extensive testing of 11 private wells, 3 municipal production wells, 3 municipal test wells, 2 containment wells on Uniroyal property, 42 test wells on Uniroyal property and 6 surface water

TABLE 4.2.3.1.A
UNIROYAL, ELMIRA WATER ANALYSES (TEST WELL 18S)*

	PCDD Degree of Chlorination								PCDF Degree of Chlorination				
DATE	1	2	3	4	5	6	7	8	4	5	6	7	8
October 6/82	ND	0.3	0.9(2)	1.3(2)	ND	ND	0.5(1)	0.5	3.3(5)	3.4(3)	ND	34(2)	13
January 10/83	-	ND	ND	5.3(2)	ND	ND	ND	ND	ND	ND	ND	ND	ND
Late/1983	-	-	-	0.1(2)	ND	ND	ND	ND	0.2(1)	0.1(1)	0.3(2)	2.4(2)	0.7

* concentrations in ng/L.
Numbers in brackets ( ) are number of isomers detected.

locations have shown that only one shallow test well, TW18S contains consistently reproducible levels of PCDDs and PCDFs (Table 4.2.3.1A). Repeat samples of other test wells were negative for PCDDs and PCDFs at the 0.02 (T_4CDDs) to 0.3 (O_8CDD) ng/L level of detection (MOE, 1983).

Implications

While some forms of PCDD and PCDF have been identified in test wells and soil in the vicinity of test well 18S, none have been identified in surface water or fish in the area. Presence of 2,3,7,8-T_4CDD in the T_4CDD fraction from TW18S has not been confirmed. Wastes from the production of 2,4,5-T are contained in drums buried on the site. The concentration of 2,3,7,8-TCDD in this waste is not known with any degree of accuracy; therefore, any estimate of total amounts is speculative.

4.2.3.2 Wood Preservation Sites

As mentioned, (Section 4.2.2.1) there are only two major users of P_5CP in wood preservation in Ontario.

I) Process Waters From Wood Preservation

The process waters from wood treatment plants are sources of chlorophenol, and possibly PCDD/PCDF pollution, if the waters are not treated either biologically or chemically (Jones, 1981). The primary source of pollution is retort condensate water (Shields, 1976). This is process water which, while flowing down walls of the retort in pressure treatment cylinders entraps P_5CP residues from previous cycles. Shields (1976) estimated that the quantity of contaminated water would be 4,500 to 9,100 L/day/retort.

Another main source is the cooling waters from barometric condensers. Shields (1976) indicates that the volume of water from this source is too large to be treated economically.

The use of an activated sludge system has been under investigation at Northern Wood Preservers Ltd., Thunder Bay, Ontario (Guo _et_ _al._, 1979). This plant, operating at near capacity, reduces phenol levels from 800 mg/L to 1 mg/L. Pentachlorophenol levels were reduced from 14.9 to 3.6 mg PCP/L; volume of wastewater discharged is 13 m^3/day.

Groundwater contamination problems at Abitibi-Price Northern Wood Preservers Ltd., Thunder Bay, Ontario, occurred through seepage of PCP-contaminated liquid from landfill into Lake Superior. This seepage problem is being contained through pumping and treating the pumped effluent through activated charcoal.

Effluents from Ontario wood preserving plants have not been analyzed for levels of PCDD/PCDF. Therefore, no estimate of PCDD or PCDF environmental input from this source can be made.

II) Sludges From Wood Preservation Plants

Boultinization, a drying process using heat and vacuum in the treatment of wood, consists of heating the wood in a hot creosote or pentachlorophenol/oil solution under vacuum. Generally, the oil temperature used is 210-220°F for a time period of 10-50 hours.

As discussed in Section 4.2.3.1 the sludge from the wood dipping tanks and the recirculating liquid involved in the Boultonization process was enriched in PCDDs and PCDFs as compared to the fresh solution. The method of disposal of these wastes may involve environmental PCDDs/PCDFs input.

Domtar Wood Preservers, Trenton, dispose of contaminated soil and mill ends at an approved landfill site. It was reported (Pruner, Pers. Comm.) that they have never disposed of their sludge from the dipping process. The underflow from their dipping process is, at present, discharged directly into the Trent River. Following mid-1984, this effluent will be treated with activated carbon prior to discharge.

High levels (2 mg/L) of PCP have been found in the Trent River at the effluent discharge from Domtar.

Abitibi Price/Northern Wood Preservers dispose of contaminated soil only at an approved site. No information could be found regarding sludge disposal.

4.2.3.3 Landfill Sites Accepting Chemicals and Other Industrial Sites Potentially Contaminated with PCDDs or PCDFs

This is a preliminary list of sites potentially contaminated following the production/use/disposal of wastes from chlorophenol products, phenoxy herbicides or PCBs. Generally no records or data are available on quantities or types of materials historically deposited at these sites.

Landfill Sites

Upper Ottawa St., Hamilton
Smithville Twp.
Tiny Twp.
Stouffville Twp.
Harwich Twp.
Gloucester Twp.
Beare Rd., Scarborough

Industrial Sites

Junction Triangle, Toronto
Tricil, Corunna
Dow, Sarnia

4.2.3.4 Incinerator Ash Disposal Sites

It has been well documented that fly ash produced from incineration of municipal garbage contains appreciable levels of PCDD/PCDF (Clement *et al.*, (in press), Ozvacic *et al.*, 1984a). Fly ash from Ontario incinerators contains 29 to 6200ng PCDD + PCDF/g. However, concentrations of PCDDs and PCDFs in bottom ash from Ontario incinerators are much lower ranging from ND (limit of detection - 0.2 to 1.0ng/g, Tosine *et al.*, 1983b) to 7.2ng/g (Ozvacic *et al.*, 1984).

Estimates of PCDD and PCDF loading from incinerator ash to landfill sites for two Ontario municipal incinerators, SWARU, Hamilton and Commissioner Street, Toronto and the main Toronto sewage sludge incinerator at Ashbridges Bay, are listed in Table 4.2.3.4A.

In calculating the loading of total PCDD and PCDF in incinerator ash to landfill sites the following assumptions were made:

a) levels of PCDDs and PCDFs in bottom ash or combined ash are either ND or extremely low; therefore, loading estimates are based on quantities of fly ash,

b) Fly ash is assumed to represent 5% of the total ash landfilled from Commissioners Street and Ashbridges Bay, and 15% of the total ash landfilled from SWARU, and

c) the average concentration of PCDDs and PCDFs in Ontario fly ash is assumed to be 137 ng/g (from Table 4.2.1.1.A).

TABLE 4.2.3.4A

ESTIMATES OF INPUT OF PCDD AND PCDF FROM INCINERATOR ASH TO LANDFILL SITES.

	SWARU	Commissioners Street	Ashbridges Bay
Municipal refuse or sewage incinerated (tonnes/year)[1]	70,800	125,752	36,500
Combined Ash delivered to landfill (tonnes/year)[1],[2]	22,460	38,199	(18,250)[4]
Total PCDD + PCDF Loading to landfill[3] (g/year)	461	785	(-)

[1] - Quantities recorded in 1982 to 1983.

[2] - Fly ash assumed to be 5% of total ash from Ashbridges Bay and 15% of total ash from Commissioners Street and SWARU

[3] - Average concentration of PCDDs and PCDFs in Ontario fly ash assumed to be 137 ng/g (Table 4.2.1.1A).

[4] - Transported to Quebec.

Commissioner Street Incinerator ash was transported in the past to Beare Landfill Site, Scarborough, after screening for metals, and is now transported to Brock West Landfill Site, Pickering.

Ash from SWARU Incinerator is removed to Glanbrook Landfill Site.

Ash from the Ashbridges Bay sewage sludge incinerator is removed by train to Noranda Mines, Quebec where it is processed for precious metal, iron and silica recovery (R.L. Pyne Metallurgical Consultants Inc., 1984). Consequently this ash is not a source of PCDDs and PCDFs to the Ontario environment at present.

No data exist on the degree of mobility of PCDD/PCDF from ashes in a landfill site.

4.2.3.5 Municipal/Industrial Sewage & Sludge Disposal

Recent estimates for 1981 to 1982 indicate that Ontario produces about 3 million m^3 of wet sludge per year as the solid waste component of municipal and industrial sewage. Assuming that the wet sludge contains 90 to 95% water this suggests an annual production of 150,000 to 300,000 dry tonnes of material. Fifty percent of this sewage sludge was incinerated at several locations namely Hamilton, Kitchener, Ottawa, Windsor, York-Durham, Highland Creek and the main Toronto unit at Ashbridges Bay. New incinerators have been proposed or are under construction at Lakeview and London. Forty percent of Ontario's sewage sludge was land filled. The rest was applied to drying beds, used for composting or used in mine tailing reclamation projects. Land application of sewage sludge is controlled by "Guidelines for Sewage Sludge Utilization on Agricultural Lands" (MOE/OMAF, 1981). Many urban or industrial sewage sludges eg. Hamilton and Elmira are unacceptable for agricultural usage due to heavy metal or PCB content and consequently must be incinerated or land filled.

An ongoing federal (Environment Canada) and provincial study is investigating the levels of PCDD/PCDF in the influent and effluent of sewage treatment plants. Table 4.2.3.5A lists the total PCDD and PCDF concentrations at the Hamilton Sewage Treatment Plant. It must be noted that the PCDDs/PCDFs found in sewage, sludge and scrubber effluents were primarily H_7CDD, but the concentration was low at 0.05 ng/L.

TABLE 4.2.3.5A

PRELIMINARY MONITORING RESULTS FOR LEVELS OF PCDD/PCDF IN HAMILTON STP (ENVIRONMENT CANADA)

Total PCDD/PCDF in raw sewage	3.18 ng/L
Total PCDD/PCDF in effluent	0.05 ng/L
Total PCDD/PCDF in sludge	18.4 ng/g
Total PCDD/PCDF in incinerator scrubber effluent	63.0 pg/L

Other sewage treatment plants serving industrial communities may also have PCDD/PCDF in the sewage.(Table 4.2.3.5B)

The significance of these low levels is not immediately apparent. No other data exist for other sewage treatment plants in Ontario.

TABLE 4.2.3.5B

PRELIMINARY RESULTS FOR LEVELS OF PCDD/PCDF IN ELMIRA STP SLUDGE (MOE)*

	Sludge
T_4CDD (0.3)**	ND
P_5CDD (1.3)	ND
H_6CDD (1.3)	ND
H_7CDD (1.3)	15000
O_8CDD (1.3)	28000
Total PCDD	43000
T_4CDF (1.3)	ND
P_5CDF (1.3)	ND
H_6CDF (1.3)	ND
H_7CDF (1.3)	ND
O_8CDF (0.8)	21000
Total PCDF	21000
Total PCDD + PCDF	45100

* - all concentrations in pg/g
( )** - average detection limits in pg/g

By comparison samples of Milorganite R (dried municipal waste treatment sludge from Milwaukee, Wisconsin sold as fertilizer) contained 33.8 to 222.4 pg T_4CDD/g, 1,412 to 1,457 pg H_6CDD/g, 7,600 to 9,700 pg H_7CDD/g and 50,000 to 60,000 pg O_8CDD/g (Lamparski et al., 1984). A sample of this material preserved since 1933 (a period predating the commercial production of chlorophenols and their products in the U.S.) had essentially the same concentrations of PCDDs.

4.2.4 NIAGARA FRONTIER AND OTHER TRANSBOUNDARY SOURCES

4.2.4.1 Niagara Frontier

In 1979 the state of New York Interagency Task Force on Hazardous Wastes identified 215 waste disposal sites in Erie and Niagara counties;

essentially, the Niagara Frontier. These sites were classified into three groups of which 36 were given a priority/rating. Priority 1 is defined as "waste disposal areas that have definitely received large quantities of hazardous wastes; remedial action may be necessary; litigation to ensure that remedial action is taken should be considered". The Priority 1 classification includes the Love Canal disposal site and the Hyde Park disposal site. Estimates of the amount of PCDDs buried in these Priority 1 disposal sites vary from 100 kg to 5,000 kg (Hallett, Pers. Comm.).

The occurrence of PCDDs in and around the Niagara River has been identified in several sources (Smith et al., 1983; Suns, 1984; O'Keefe et al., 1983). These investigations demonstrate conclusively that parts of the Niagara River watershed are contaminated with PCDDs. Smith et al. (1983) sampled the sediment in storm sewers in and around the Love Canal chemical disposal site. The storm sewers drained from the disposal site directly or indirectly through Cayuga Creek to the Niagara River. These samples show that 2,3,7,8-T_4CDD is gaining access to the Niagara River through the sewers. Additionally, O'Keefe et al. (1983) have established the occurrence of 2,3,7,8-T_4CDD in four species of fish taken from Cayuga Creek, which is in the immediate vicinity of the area studied by Smith et al. (1983). O'Keefe et al. (1983) found concentrations of 2,3,7,8-T_4CDD in the fish ranging from 12 to 87 pg/g.

Limited sampling and analysis of surface water from the Niagara River near the extensive S-Area chemical landfill site known to contain large quantities of 2,3,7,8-T_4CDD and of other PCDD and PCDF wastes has been carried out by Environment Canada.

Surface water concentration distributions are given in Table 4.2.4.1A below. These data apply to a very localized area in the Niagara River, (Environment Canada, 1983).

TABLE 4.2.4.1.A
CONCENTRATIONS OF PCDDs* AND PCDFs IN NIAGARA RIVER WATER NEAR THE HYDE PARK DUMP SITE, NIAGARA FALLS, N.Y. 1983

	ng/L
PCDD Concentration Range	0.001 - 0.20
PCDF Concentration Range	0.005 - 0.05

Environment Canada (1983)
* presence of 2,3,7,8-T_4CDD not confirmed.

4.2.4.2 Dow, Midland, Michigan

Effluents from Dow Chemical, Midland, Michigan as well as fish from the Tittabawassee River near Dow have been found to contain levels of PCDD/PCDF (Table 4.2.2.2.2A). However, data from fish caught in Lake Huron (Hoering, 1983) indicate no detectable 2,3,7,8-T_4CDD (detection 10 ppt) for whitefish. Ministry of the Environment data for 1980 fish collections of lake trout, rainbow trout and walleye (18 samples) indicate no detectable levels of 2,3,7,8-T_4CDD at detection limits of 10 ppt.

Therefore, contribution of Dow Chemical effluents to the Ontario environment can be assumed to be minimal.

TABLE 4.2.4.2A

LEVELS OF T_4CDD IN FISH FROM TITTABAWASSEE RIVER AND SAGINAW BAY, LAKE HURON (NEAR DOW).

Type of Fish	Location	Range of 2,3,7,8-T_4CDD Concentration (pg/g)	No. of Samples	Reference
Carp	Tittabawassee River	17-301	10	Kaczmar *et al*. 1983
Sucker	Tittabawassee River	64	1	"
Carp	Saginaw River	nd-319	6	"
Channel Catfish	Tittabawassee	157	Not Known	Harless *et al*. 1982
Carp	Tittabawassee	55	Not Known	"
Yellow Perch	Tittabawassee	13	Not Known	"

4.2.4.3 St. Clair River/Detroit River

Gull eggs from Fighting Island, Detroit River have been found to contain levels of 40 ng/kg 2,3,7,8-T_4CDD. This concentration is above background, as measured from other locations of the Great Lakes (Canadian Wildlife Service, 1980).

Also, low levels (80-90 pg/g) of H_7CDD and O_8CDD have been detected in two core samples of soils from Fighting Island (Windsor Star, 1984).

Walleye caught in Lake St. Clair in 1979 showed no PCDD levels (U.S. Fish and Wildlife Service, 1979). However, PCDF levels in the same fish were 4 pg T_4CDF/g, 4 pg P_5CDF/g, 0.7 pg H_6CDF/g, 0.7 pg H_7CDF/g and 3 pg O_8CDF/g.

From these data, it appears that the St. Clair and Detroit River areas may have some contamination by PCDF, the magnitude of which cannot be assessed at this time.

4.2.4.4 Ontario Water Utilities

Several programs have recently been completed examining PCDD/PCDF in various municipal water supplies. Assorted water samples (raw and treated) were collected from 13 widely distributed water treatment plants. The results, detailed in Table 4.2.4.4A show that at a detection level of 1 ng/L total 2,3,7,8-T_4CDD, none were detected.

In the two-year period from June, 1980 to September, 1982 two sets of water samples were analyzed from nine water works in Western Lake Ontario. The first set of 91 samples had a detection limit of 1 ng/L total 2,3,7,8-T_4CDD (Table 4.2.4.4B). The second set of 52 samples had a detection limit of 0.25 ng/L total 2,3,7,8-T_4CDD (Table 4.2.4.4C). 2,3,7,8-T_4CDD was not detected in any of these 143 samples.

TABLE 4.2.4.4A

ANALYSIS OF WATER SAMPLES FOR TOTAL 2,3,7,8-T_4CDD JANUARY 1980 - JANUARY 1981

Location	Sample Type	Number of Samples Analyzed	Result
Thunder Bay	raw	1	ND*
	treated	1	ND*
Sault Ste. Marie	raw	2	ND*
Sudbury (W.T.P.-North Channel)	raw	2	ND*
Parry Sound	raw	1	ND*
Lambton	raw	1	ND*
	treated	1	ND*
Essex County (Union Water System)	raw	1	ND*
	treated	1	ND*
Grimsby	raw	1	ND*
Hamilton	raw	1	ND*
Metro Toronto	raw	2	ND*
Whitby	raw	1	ND*
Belleville	raw	1	ND*
Sidney Twp.	raw	1	ND*
Kingston	raw	2	ND*
	Total	20	

* Detection Limit - 1 ng/L, Total 2,3,7,8-T_4CDD

Source: Internal Memo. R.D. Smillie to G.C. Ronan, Analysis of Water Samples for Dioxin. January 21, 1981

TABLE 4.2.4.4B
ANALYSIS OF WATER SAMPLES FOR TOTAL 2,3,7,8-T_4CDD (WESTERN LAKE ONTARIO)
JUNE 1980 - SEPTEMBER 1981

Location	Sample Type	Number of Samples Analyzed	Result
Niagara Falls	raw	14	ND*
	treated	12	ND*
Niagara-on-the-Lake	raw	13	ND*
	treated	11	ND*
St. Catherines	raw	13	ND*
	treated	13	ND*
Fort Erie	raw	14	ND*
Lincoln/Vineland	-	1	ND*
	Total	91	

* Detection Limit - 1 ng/L, Total 2,3,7,8-T_4CDD
Source: H. Tosine Internal Communication 1983 to G.H. Mills, Director W/C Region.

TABLE 4.2.4.4C
ANALYSIS OF WATER SAMPLES FOR TOTAL 2,3,7,8-T_4CDD (WESTERN LAKE ONTARIO)
SEPTEMBER 1981 - SEPTEMBER 1982

Location	Sample Type	Number of Samples Analyzed	Result
Niagara Falls	raw	9	ND*
	treated	9	ND*
Niagara-on-the-Lake	raw	10	ND*
	treated	8	ND*
St. Catharines	raw	3	ND*
	treated	3	ND*
Fort Erie	raw	7	ND*
Grimsby	-	1	ND*
Port Colborne	-	1	ND*
Lincoln/Beamsville	-	1	ND*
	Total	52	

* Detection Limit - 0.25 ng/L, Total 2,3,7,8-T_4CDD
Source: Internal Memo Tosine - Mills, 1983

Re-analysis of the 1982 samples was reported in January, 1983. The results indicated that at more sensitive detection limits of 0.005 ng/L traces of total T_4CDD were occurring in raw water only. In the treated water total T_4CDD was not detected. Table 4.2.4.4D shows the results for the only three samples in which T_4CDD was detected.

TABLE 4.2.4.4D

ANALYSIS OF WATER SAMPLES FOR TOTAL T_4CDD (WESTERN LAKE ONTARIO) COMPARISON OF MINISTRY OF ENVIRONMENT AND HEALTH AND WELFARE, CANADA DATA

Location	Sample Type	Health & Welfare Canada High Resolution GC/MS (ng/L)	Ministry of Environment Low Resolution GC/MS
Niagara-on-the-Lake	raw	0.010	ND**
	treated	ND+	ND*
Niagara Falls	raw	0.017	ND**
	treated	ND+	ND*
St. Catharines	raw	0.028	ND**
	treated	ND+	ND*

** ND = "traces" were detected at less than 0.050 ng/L.
ND* = none detected to 0.250 ng total T_4CDD/L
ND+ = none detected to 0.005 ng total T_4CDD/L

Source: Internal Memo H. Tosine - G.H. Mills, Director Western Region; Analysis of Niagara Area Waters for TCDD January 31, 1983.

Using a more sensitive methodology (0.01 ng/L detection limit), an extensive survey of 15 Lake Ontario waterworks was done in the summer of 1983 (Tosine et al., 1984). Twenty-six samples each of raw and treated drinking water were collected. No PCDD or PCDF was detected in any of the treated drinking water samples. Only the O_8CDD congener was detected at levels only 3 to 4 times the detection limit in the raw water supply of St. Catharines and Lakeview waterworks.

Since no PCDDs or PCDFs have been detected in treated drinking water at parts-per-quadrillion limits, potable water from conventional water treatment facilities in Ontario can be ruled out as a source of exposure of these substances. Occasional monitoring of drinking water from treatment plants, however, should be continued to ensure this high standard of water quality is maintained.

4.2.5 SUMMARY OF PCDDs AND PCDFs ENTERING THE ONTARIO ENVIRONMENT

Estimates of inputs of PCDDs and PCDFs from combustion sources, chemical manufacturing and use, waste disposal and transboundary sources to the atmospheric, terrestrial and aquatic components of the Ontario environment are summarized in table 4.2.5A. Sources are listed in order of total quantity of PCDDs and PCDFs input to the environment. Quantitative assessment of PCDD and PCDF inputs from the sources listed is limited by the available data.

Table 4.2.5A indicates that chemical products are a major source of PCDDs and PCDFs. Chlorinated phenols form the major portion of the total potential input from this source. It is important to note that no 2,3,7,8-T_4CDD is detectable in chemical products currently used in Ontario. Phenoxy herbicides only contain trace amounts of the di- and trichlorinated PCDD congeners. At least 70% of the PCDDs and PCDFs in the major chlorinated phenols (P_5CP, T_4CP) are the octachlorinated forms. These PCDDs and PCDFs are 10^4- to 10^5-fold less toxic than 2,3,7,8-T_4CDD (Section 3.6.7). Aroclors (PCBs) may contain significant quantities of toxic tetra- and pentachlorinated PCDFs; however, these fluids are enclosed and currently scheduled for destruction.

Combustion sources represent the next major source of PCDDs and PCDFs to the Ontario environment. Assessment of input is more complicated because of all the potential combustion sources described in Section 4.2.1. Only municipal waste and sewage sludge incinerators have been adequately investigated in Ontario.

Considerable quantities of fossil fuels, as well as treated and untreated wood are incinerated in Ontario. Forest fires may also contribute significant quantities of PCDDs and PCDFs to the Ontario environment.

Transboundary sources are unquantifiable at present but reflect the industrial activity of the east central part of North America. Large chemical waste sites on the Niagara Frontier represent a major source of PCDDs and PCDFs to Lake Ontario.

Sewage sludge is the endpoint of much human, municipal and industrial activity. The predominant PCDD and PCDF congeners identified so far are the hepta- and octachlorinated forms.

In order to put a different perspective on the order of significance of the sources/inputs listed in Table 4.2.5A these estimated quantities of total PCDD plus PCDFs were recalculated as 2,3,7,8-T_4CDD toxic equivalents using the relative toxicity factors in Table 3.6.7C. The resulting estimates of quantities of PCDDs and PCDFs entering the Ontario environment as 2,3,7,8-T_4CDD toxic equivalents are shown in Table 4.2.5B.

This analysis indicates that combustion sources are the major source of concern. While chemical products may introduce the largest quantity of total PCDD plus PCDF into the environment, the high proportion of less toxic congeners ameliorates the potential toxicity of this source.

Both attempts to estimate the relative significance of various sources of PCDDs and/or PCDFs are flawed by the substantial data gaps indicated in these analyses eg. items 1D, 1E, 2B, 3A, 3B and 4 in both Tables 4.2.5A and B.

TABLE 4.2.5.A

Page 1 of 4

SUMMARY OF ESTIMATED QUANTITIES OF PCDDs AND PCDFs ENTERING THE ONTARIO ENVIRONMENT

SOURCE	USE/ORIGIN	ESTIMATED CURRENT QUANTITY	RELEASE TO ENVIRONMENT	REFERENCE SECTION
1. CHEMICAL PRODUCTS (No 2,3,7,8-T_4CDD in chemicals in current use)				
A. Phenoxy Herbicides (2,4-D, MCPA, Dicamba)	- sprayed on crops/forests	4 to 12g PCDDs annually	Open - 90% on crop/10% to soil or water _ fraction of application rate	4.2.2.2
B. Chlorinated Phenols (P_5CP, T_4CP, T_3CP)	- primary wood treatment	503 kg PCDD + PCDF annually (140 kg or 30% is H_6CDD, H_6CDF, H_7CDD and H_7CDF)	Variable - some local - due to treatment and waste disposal - rest distributed in product	4.2.2.1
	- pole retreatment (Based on Ontario Hydro usage)	8 - 10kg PCDD +PCDF annually (2 - 3kg is H_6CDD, H_7CDD, H_6CDF, and H_7CDF)	Variable - injected into soil around utility poles _ limited movement in soil near poles	4.2.2.1
C. Polychlorinated Biphenyls (PCBs)	- in electrical equipment in use	14.6 kg PCDFs (not an annual input)	Closed - Some local release if leakage or fires	4.2.2.4
	- in storage for disposal	3.4 kg PCDFs (not an annual input)	Closed - ditto	
D. Chemical Wastes (A, B and C)	- industrial on-site processes and waste disposal practices	Unknown (ng/L levels in leachate)	Variable - mainly in leachate from industrial sites or dumps accepting chemical waste	4.2.3

TABLE 4.2.5A

SUMMARY OF ESTIMATED QUANTITIES OF PCDDs AND PCDFs ENTERING THE ONTARIO ENVIRONMENT

	SOURCE	USE/ORIGIN	ESTIMATED CURRENT QUANTITY	RELEASE TO ENVIRONMENT	REFERENCE SECTION
1.	CHEMICAL PRODUCTS (continued)				
	E. Commercial/ Domestic Products	- in home use of: 1) wood preservatives or treated wood products 2) lawn and garden weed control products 3) disinfectants and dental care products	Unknown	Variable - dermal contact with 1), 2) or 3) is major source of contamination	4.2.2.5
2.	COMBUSTION				
	A. Municipal Refuse/ Sewage Sludge	(refuse reduction and energy from waste programs)			
		(a) airborne - emitted particles and stack gas	28 to 32 kg PCDDs + PCDFs annually	Open - mainly dispersion zones around sources	4.2.1.1 and 4.2.1.7
		(b) precipitated fly ash and bottom ash	1 to 5 kg PCDD and PCDF annually	Variable - most placed in landfill potential for movement in leachate	4.2.3.4

Page 3 of 4

TABLE 4.2.5A

SUMMARY OF ESTIMATED QUANTITIES OF PCDDs AND PCDFs ENTERING THE ONTARIO ENVIRONMENT

SOURCE	USE/ORIGIN	ESTIMATED CURRENT QUANTITY	RELEASE TO ENVIRONMENT	REFERENCE SECTION
2. COMBUSTION (continued)				
B. All Other Combustion Sources	- fossil fuel combustion for power; - residential wood combustion; - industrial combustion of treated and untreated wood wastes - hospital and apartment building waste incineration; - industrial waste solvent combustion; - forest fires; - motor vehicles; - tobacco; - PCB fires	100 to 200 kg PCDD + PCDF annually based on Federal estimates (atmospheric, residual ash and dustfall)	Open - Dispersion zone depends on magnitude or height of source	4.2.1.2 to 4.2.1.6

Page 4 of 4

TABLE 4.2.5A

SUMMARY OF ESTIMATED QUANTITIES OF PCDDs AND PCDFs ENTERING THE ONTARIO ENVIRONMENT

	SOURCE	USE/ORIGIN	ESTIMATED CURRENT QUANTITY	RELEASE TO ENVIRONMENT	REFERENCE SECTION
3.	TRANSBOUNDARY				
	A. Airborne	- particulate matter (mainly from fossil-fuelled power plants - vapour phase (PCDFs in PCBs)	Unknown	Open - very diffuse-related to movement of air masses over continent	4.3.1
	B. Waterborne	- industrial processes and waste sites along lower Great Lakes and connecting waterways, e.g. Niagara frontier	2 to 20 kg PCDD + PCDF estimated to flow annually into Lake Ontario (Environment Canada)	Open - very dilute-associated with large movement of water	4.2.4.1 to 4.2.4.4
4.	SEWAGE	Human, municipal and some industrial wastes 150,000 to 300,000 dry tonnes produced per year	Unknown (ng/g levels in sludge)	Variable - 60% incinerated or applied to land with resulting contributions to the atmosphere, soil or foodstuffs 40% is landfilled - potential for movement in leachate	4.2.3.5

TABLE 4.2.5.B

SUMMARY OF ESTIMATED QUANTITIES OF PCDDs AND PCDFs ENTERING THE ONTARIO ENVIRONMENT AS 2,3,7,8-T_4CDD TOXIC EQUIVALENTS ([a])

SOURCE	ESTIMATED CURRENT QUANTITY
1. CHEMICAL PRODUCTS (No 2,3,7,8-T_4CDD in chemicals in current use)	
A. Phenoxy Herbicides (2,4-D, MCPA, dicamba)	2 - 7 x 10^{-5} kg annually[b]
B. Chlorinated Phenols (P_5CP, T_4CP, T_3CP)	4 kg annually
C. Polychlorinated Biphenyls (PCBs)	5 kg (not an annual input)
D. Chemical Wastes (A, B and C)	Unknown
E. Commercial/Domestic Products	Unknown
2. COMBUSTION	
A. Municipal Refuse/Sewage Sludge	8 - 10 kg annually
B. All Other Combustion Sources	20 - 50 kg annually
3. TRANSBOUNDARY	
A. Airborne	Unknown
B. Waterborne	<2 - 20 kg annually
4. SEWAGE	Unknown

(a) - Refer to Table 4.2.5A for detailed description of patterns of use and environmental release

(b) - 2,3,7,8-T_4CDD toxic equivalent calculations based on following information:
source 1.A - Table 4.2.2.2B
source 1.B - Table 4.2.2.1A
source 1.C - Table 4.2.2A
source 2.A - Table 4.2.1.1E
source 2.B - assumed toxic equivalent factor was intermediate between fly ash and stack emissions

4.3 ENVIRONMENTAL TRANSPORT, DISTRIBUTION AND FATE

4.3.1 ATMOSPHERIC TRANSPORT

4.3.1.1 Long Range Transport in the Atmosphere

According to Somers and Douglas (1983) the long-range transport of PCDDs and PCDFs has been debated for a number of years, owing primarily to their presence in pesticides, PCBs, food, air and water. There is, at this time, no direct evidence (atmospheric monitoring) linking the movement of PCDDs and PCDFs into Ontario via long-range transport. There is evidence, however linking the presence of PCDDs and PCDFs in sediments from Lake Siskiwit on Isle Royale in Lake Superior to the atmospheric transport of these compouonds (Czuczwa et al., 1984).

4.3.1.2 Chemical Stability in the Atmosphere

Townsend (1980,1982,1983) has conducted a number of studies designed to estimate the stability of PCDDs in various types of sample materials, based on isomer group ratios normalized to O_8CDD. According to his redox hypothesis, the newly created PCDDs immediately start to oxidize and simultaneously undergo isomer group redistribution within the (combustion) source. The PCDDs which have been found on airborne particulates suggest that the redox process may continue in the atmosphere to create a low-level background. This process is estimated to occur within the first 300 meters of the emission source and beyond this distance the PCDD laden particulates mix into and help create the environmental background. In another study (Lamparski et al. 1982) it is suggested that particulate disintegration and subsequent microparticulate agglomeration may transport and mix low volatility PCDDs in the atmosphere.

Photodegradation must also be considered as a factor affecting the ultimate fate of PCDDs and PCDFs emitted into the atmosphere either in a particulate or aerosol form. Although there have been no controlled studies designed to address this concept from an atmospheric transport aspect, the effectiveness of ultra-violet light in degrading these compounds in the presence of a hydrogen donor is well documented (Section 4.3.2.3).

4.3.1.3 Atmospheric Monitoring

Although the mechanisms affecting stability and transport of PCDDs in the atmosphere are not fully known there is a growing base of environmental monitoring data which should prove useful in evaluating these concepts. For this reason and in order to provide an update of atmospheric monitoring results, Table 4.3.1.3A has been developed based on the available literature. No attempt will be made to consolidate these results due to their limited number and to the wide range in the atmospheric component measured (dust fall, suspended particulates, gaseous).

4.3.2 DEPOSITION, FATE AND MOBILITY IN THE TERRESTRIAL ENVIRONMENT

The deposition of PCDDs to soil surfaces has been the subject of extensive investigations following the accidental release of 2,3,7,8-T_4CDD at Seveso, Italy; the disposal of still bottom contaminated waste oils for dust control purposes in Missouri;

and the testing and storage of T_4CDD contaminated herbicides which were used by the U.S. Air Force for war-related defoliation practices in South-East Asia.

Table 4.3.2.1A summarizes the results of the analytical findings and, in addition, presents a number of other published results which are of use in establishing a background range for a few of the PCDD and PCDF isomer groups detected in soils.

4.3.2.1 Deposition to Soils and Plants from Atmospheric Sources

Although considerable progress has been made in analytical detection techniques since the 1978 report by Dow Chemical Co., no other report providing this type of data for PCDD concentrations in urban and rural soils has been published in the open literature. Thus the findings of the Dow investigation which revealed less than detectable concentrations (low pg/g range) of the T_4 and H_6 isomer groups and levels between 0.02 - 0.05 and 0.1 - 0.2 ng/g of H_7- and O_8CDD, respectively, for a rural control area, represent the best available background estimates. The other soil data reported by DOW also provide some indication of urban and major metropolitan PCDD soil concentrations, although in each case the sampling was conducted in the vicinity of either powerhouse or incineration (not identified) sources. However, as no firm trends were apparent, with respect to the proximity to the various sources (Crummett, 1981), the data probably reflect typical urban soil levels. Again, the T_4CDD results generally fall in the low pg/g range (most not detected) while the other chlorinated isomer groups generally were higher than corresponding rural area results, with values for

Page 1 of 2

TABLE 4.3.1.3A

ATMOSPHERIC MONITORING RESULTS

REFERENCE	SOURCE/ DESCRIPTION	MONITOR DESCRIPTION/ LOCATION	UNITS	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
DOW Chemical Co. (1978)	Midland Manufacturing Plant	ambient air near plant	ug/m^3	<0.1									
Di Domenico *et al.* (1980a)	ICMESA Factory, Seveso, Italy	dustfall jars	ng/g	0.06–2.1									
		hi-vol filters	ng/g	0.17–0.50									
		pooled hi-vols. June 1977											
		Site Dust Conc. (mg/m^3)	ng/g (pg/m^3)										
		A 0.139		0.432(0.06)									
		B 0.115		0.504(0.058)									
		C 0.119		<0.168(<0.02)									
		pooled dust fall results (Mar.–Oct, 1978)	ng/m^2/day	<0.01–0.23									

TABLE 4.3.1.3A

ATMOSPHERIC MONITORING RESULTS

REFERENCE	SOURCE/ DESCRIPTION	MONITOR DESCRIPTION/ LOCATION	UNITS	PCDD					PCDF				
				T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
Lamperski & Nestrick (1980)	Urban U.S.A. (St. Louis, MO)	urban particulate (NBS Sample)	ng/g	0.28(0.12)*		2	16** 18***	210					
				* 2,3,7,8-TCDD and 4 other isomers ** 1,2,3,4,6,7,9-H_7CDD *** 1,2,3,4,6,7,8-H_7CDD									
Kleopfer *et al.* (1983)	Chemical plant - Verona, Missouri	florisil adsorption tube -site of T_4CDD storage drum	ng/m^3	<1									
Olie *et al.* (1983)	Municipal Waste Incinerator - Zaanstad, The Netherlands	suspended particulate											
		0.9 Km N.E.	pg/m^3	1.5	3.4	4.0	5.2	13.3	5.0	4.8	5.2		4.8
		2.0 Km N.E.		0.1	0.5	2.0	2.1	0.9	0.4	0.6	2.1		0.4

TABLE 4.3.2.1A

Page 1 of 6

SUMMARY OF PCDD AND PCDF LEVELS IN SOILS

REFERENCE	SOURCE/ DESCRIPTION	SAMPLE DESCRIPTION/ LOCATION	PCDD				PCDF
			T_4	H_6	H_7	O_8	T_4
			(ng/g)				
Commoner & Scott (1976)	Horse Arenas - Missouri sprayed with T_4CDD contaminated waste oils	Arena A - sample depth unknown (soon after spraying) at 16" depth	31,000-33,000*				
		-(39 mo. after spraying and 28 mo. after surface replacement) nearby pasture (0-4 in.)	ND* TR*				
		Arena B -soil excavated from area and exposed outside for 15 mos. (3 sites in pile)	440*, 850*, 380*				
		Arena C - depth of 10-14 in. -(38 mos. after3 mos. after soil from top 12 in. removed)	ND*				
		Arena D - (39 mos. after spraying)					
		- 3 in. depth	120*				
		- 12 in. depth	ND*				
		- 18 in. depth	ND*				

TABLE 4.3.2.1A

SUMMARY OF PCDD AND PCDF LEVELS IN SOILS

REFERENCE	SOURCE/ DESCRIPTION	SAMPLE DESCRIPTION/ LOCATION	PCDD				PCDF
			T_4	H_6	H_7 (ng/g)	O_8	T_4
Dow Chemical Co. (1978)	Dow Manufacturing Plant, Midland, Michigan	Vicinity of Dow plant (1 cm depth)	33	280	3200	20500	
			15	40	470	2500	
			118	120	650	6300	
			34	280	240	11700	
			1.1	7	70	490	
	Control area, Gaylord, Michigan	Rural area (no source) (depth unknown)	ND(<0.003)	ND(<0.03)	0.03	0.10	
			ND(<0.005)	ND(<0.05)	0.05	0.17	
			ND(<0.005)	ND(<0.02)	0.02	0.16	
			ND(<0.007)	ND(<0.005)	ND(<0.03)	ND(<0.03)	
			ND(<0.007)	ND(<0.03)	0.03	0.11	
	Powerhouses Lansing and East Lansing, Michigan	Urban Area 1 cm depth, 600 ft. ENE	ND(<0.01)	1.2	1.6	2.0	
		900 ft. ENE	ND(<0.01)	ND(<0.04)	0.23	0.96	
		1500 ft. ENE	ND(<0.007)	0.03	0.30	2.0	
		1 cm depth, 600 ft. NE	ND(<0.005)	ND(<0.05)	ND(<0.03)	0.05	
		300 ft. NE	ND(<0.009)	ND(<0.04)	0.035	0.20	

Page 3 of 6

TABLE 4.3.2.1A

SUMMARY OF PCDD AND PCDF LEVELS IN SOILS

REFERENCE	SOURCE/ DESCRIPTION	SAMPLE DESCRIPTION/ LOCATION	PCDD				PCDF
			T_4	H_6	H_7	O_8	T_4
			(ng/g)				
Dow Chemical Co. (1978) (Cont'd)	Incinerators, Chicago, Illinois	Major Metropolitan Area					
		1 cm depth, 100 ft. NE	ND(<0.02)	ND(<0.03)	0.14	0.41	
		200 ft. NE	ND(<0.01)	0.03	0.24	1.0	
		400 ft. NE	0.03	0.31	3.3	22.0	
		1000 ft. NE	ND(<0.02)	0.12	1.4	8.5	
		1cm depth, 100 ft. NE	0.006	0.14	0.85	3.2	
		200 ft. NE	0.005	0.04	0.36	1.4	
		400 ft. NE	0.005	0.09	0.96	6.0	
		1000 ft. NE	ND(<0.006)	0.02	0.10	0.35	
Wipf et al. (1982)	TCP Factory - Seveso, Italy (Accidental release)	Zone A (close to plant)	55*(highest)				
		Zone A_2/A_3	20*(highest)				
		Zone A/B (all samples 0-7 cm depth)	0.15*(highest)				
VanNess et al. (1980)	Trichlorophenol Manufacturing Site (U.S.A.)	unknown	0.02 - 559				

Page 4 of 6

TABLE 4.3.2.1A

SUMMARY OF PCDD AND PCDF LEVELS IN SOILS

REFERENCE	SOURCE/ DESCRIPTION	SAMPLE DESCRIPTION/ LOCATION	PCDD				PCDF
			T_4	H_6	H_7	O_8	T_4
			(ng/g)				
Baker & Matheson (1981)	Wood Preserving Plant, Truro, Nova Scotia	Soil (0-15 cm)					
		- on property		10**	100**	567	
		- off property (river-bank)				ND(<0.01)	
	Wood Preserving Plant, Newcastle, New Brunswick	- at plant entrance				16	
		- on property		100**	1000**	1500	
		- off property				280	
Hryhorczuk et al. (1981)	Wire reclamation Incinerator - Illinois	0-5 cm: 50 m E	0.021				0.23
		0-3 cm: 1 Km NE	ND(<0.003)				ND(<0.003)
Kleopfer et al. (1983)	Hexachlorophene Waste Drum Disposal Site, Verona, Missouri	depth not given	74* (highest)				
Young (1983)	Herbicide Agent Orange Test Spray Site, Eglin Air Force Base - Florida	Grid 1 - Sprayed from 1961 - 1964 - samples collected 1974					
		0-2.5 cm	0.15*				
		2.5-5.0 cm	0.16*				
		5.0-10.0 cm	0.7*				
		10.0-15.0 cm	0.44*				
		15.0-90.0 cm	ND*(<0.01)				

TABLE 4.3.2.1A

Page 5 of 6

SUMMARY OF PCDD AND PCDF LEVELS IN SOILS

REFERENCE	SOURCE/ DESCRIPTION	SAMPLE DESCRIPTION/ LOCATION	PCDD T_4	PCDD H_6	PCDD H_7 (ng/g)	PCDD O_8	PCDF T_4
Young (1983) (Cont'd)	Herbicide Test Spray Site, (Agent Orange) Eglin Air Force Base - Florida	Grid I 0-15 cm (Range)	0.01-1.5*				
		(Ave.)	0.325*				
		Grid II 0-15 cm (Range)	0.01-0.47*				
		(Ave.)	0.115*				
		Grid IV 0-15 cm (Range)	0.01-0.15*				
		(Ave.)	0.03*				
Young et al. (1983)	Herbicide Storage Sites, (Agent Orange) USAF Gulfport, Mississippi and Johnson Island	Gulfport, Ave. 6 old spill sites (0-8 cm) (Aug., 1977)	240* ± 270				
		Johnson Island, Ave. 8 old spill sites (0-8 cm) (Aug., 1977)	73* ± 73				
		Ave. 27 old spill sites (0-8 cm), (Jan., 1978)	29* ± 48				
		Ave. 27 old spill sites (0-8 cm), (Oct., 1978)	37* ± 58				
		Ave. 27 old spill sites (0-8 cm), (Aug., 1979)	41* ± 49				

TABLE 4.3.2.1A

Page 6 of 6

SUMMARY OF PCDD AND PCDF LEVELS IN SOILS

REFERENCE	SOURCE/ DESCRIPTION	SAMPLE DESCRIPTION/ LOCATION	PCDD				PCDF
			T_4	H_6	H_7	O_8	T_4
			(ng/g)				
Young *et al.* (1983)		**Gulfport** Recent (1977) spill site (Sampled June, 1979)					
		soil 0-8 cm(surface layers)	325*				
		8-16 cm (above hardpan)	340*				
		16-24 cm (within hardpan)	21*				
		24-32 cm (" ")	ND*(<0.48)				
		Johnson Island Recent (1977) spill site (Sampled Aug., 1979)					
		Site 10 0- 8 cm	119*				
		8-16 cm	44*				
		16-24 cm	49*				
		Site 11 0- 8 cm	116*				
		8-16 cm	11*				
		16-24 cm	8*				

* - 2,3,7,8-T_4CDD analysis (all others total homologue).

** - estimated concentration.

H_6-, H_7- and O_8CDD in the range of 0.02 - 1.2, 0.03- 3.3 and 0.05 - 22 ng/g, respectively. Since the sampling depth was in all cases one centimeter these data may underestimate the levels in sub-surface soil layers which would not be exposed to the degradative action of ultra-violet light.

In Ontario, a soil sampling survey was initiated in 1983 in the vicinity of a major municipal refuse incinerator in Hamilton to determine if PCDDs and PCDFs identified in stack emissions (Table 4.2.1.1C) had accumulated in surface (0-5cm) soils. The results, which included 11 sample sites in the vicinity of the incinerator as well as up-wind urban and remote control locations are shown in Table 4.3.2.1B.

All 14 of the soil samples had detectable quantities of at least one of the five PCDD (T_4-O_8) isomer groups whereas eight samples contained detectable levels of one or more PCDF congeners. Only one site, at 1570 m SSE, had a measurable quantity (0.007 ng/g) of T_4CDD in the soil. The highest PCDD concentration was 3.5 ng/g O_8CDD detected 1250 m SW; however, a similar level of 3.2 ng/g O_8CDD was found in the soil at one of the urban control sites (5570 m ESE) well remote from the plant. The soil at the remote/rural control site which was located at the bottom of a 50 m deep ravine approximately 22 km N of the plant contained 0.810 ng/g O_8CDD. The levels of the other PCDD and PCDF congeners in soil from the 11 sites around the plant generally were within the range detected in the urban control soils. No concentration gradients for any of the PCDDs or PCDFs were apparent when compared with distance or direction

TABLE 4.3.2.1B

PCDD AND PCDF CONCENTRATION IN SURFACE SOIL* COLLECTED IN THE VICINITY OF A MUNICIPAL REFUSE INCINCERATOR, HAMILTON, ONTARIO, 1983

Distance & Direction from Incinerator	PCDD (ng/g)					PCDF (ng/g)				
	T_4	P_5	H_6	H_7	O_8	T_4	P_5	H_6	H_7	O_8
70 m W	nd	nd	nd	0.096	0.110	0.071	nd	nd	nd	0.009
880 m SE	nd	nd	nd	nd	0.120	0.043	nd	nd	nd	nd
1100 m SW	nd	nd	nd	nd	0.310	nd	nd	nd	nd	nd
1260 m NE	nd	nd	nd	0.150	nd	nd	nd	nd	nd	nd
1260 m SW	nd	0.580	0.170	0.390	3.50	nd	nd	nd	0.180	nd
1570 m SSE	0.007	nd	nd	0.042	0.140	nd	nd	nd	nd	nd
2020 m SSE	nd	nd	nd	0.042	1.30	nd	nd	nd	nd	0.005
2100 m SW	nd	nd	nd	nd	0.075	nd	nd	nd	nd	nd
2140 m NE	nd	nd	nd	nd	0.050	nd	nd	nd	nd	nd
2380 m E	nd	nd	nd	nd	1.00	nd	nd	nd	nd	0.033
2480 m SW	nd	nd	nd	0.270	0.690	nd	nd	nd	0.150	nd
Urban Control										
4450 m SW	nd	nd	nd	0.005	0.940	0.009	0.006	nd	nd	nd
5570 m ESE	nd	nd	nd	0.097	3.20	0.068	nd	nd	nd	0.081
Rural Control 22000 m N	nd	nd	nd	nd	0.810	nd	nd	nd	nd	nd
Detection Limit: (ng/g)	0.0003	0.0013	0.0013	0.0013	0.0013	0.0013	0.0013	0.0013	0.0013	.008

* - Surface soil (sodded), 0-5 cm depth, **(dry weight basis).

nd - Not detected (less than the analytical detection limits).

relative to the incinerator (Table 4.3.2.1B.) or with their proximity to the calculated maximum ground level concentration (1182m).

The soil data indicate that PCDDs and PCDFs are present in trace quantities in urban soil in Hamilton and in the case of O_8CDD, in undisturbed soil remote from urbanization.

In another Ontario study (Clement, 1984) concentrations of PCDDs and PCDFs (T_4-O_8) were determined in the 0 - 6 cm. soil layer at the UniRoyal plant in Elmira where phenoxyacetic acids were produced and stored. In all, 5 sites were sampled with the total (T_4-O_8) PCDD and PCDF concentrations varying from 0.78 - 6.1 ng/g and 0.076 - 13.0 ng/g, respectively. On an isomer group basis, detection of PCDDs was confined to the H_6-, H_7- and O_8CDDs with the concentrations generally increasing with each successive increase in chlorination. In the case of PCDFs isomer group detection and concentration was more variable with the only consistency among the 5 samples being the presence of H_7CDF in each. In comparing the concentration of total PCDDs with the total PCDD plus PCDFs the percent composition also was found to be quite variable ranging from 22 - 94%. As this site was used for the production and storage of phenoxyacetic acids the contribution from atmospheric sources is unknown.

In an Illinois study (Hryhorczuk et al., 1981) the rural control value for the 0-3 cm depth provides further evidence that background T_4CDD levels lie below the detection limit of 3 pg/g. This also is true for T_4CDF. On the basis of this limited sampling program airborne T_4CDD and T_4CDF deposition to surface soils has not been demonstrated in the immediate vicinity of this incinerator.

The soil analysis findings for the U.S. Air Force studies, the Seveso accident and the Missouri horse arenas (Table 4.3.2.1A) provide additional evidence of significant PCDD deposition and accumulation in surface soils. These data also give some indication of the limited mobility and the persistence of these compounds in soils, an aspect which is covered in Section 4.3.2.3.

Although actual residue data on the accumulation of T_4CDD in soil following the application of 2,4,5-T at recommended doses have not been reported, Norris et al. (1977) have estimated the input to soil (0-15 cm depth) assuming the T_4CDD persistence characteristics reported by Crosby and Wong (1977) and Kearney et al. (1973). These estimated values ranged from 0.001pg/g immediately after application down to 0.0005 pg/g 52 days later.

Atmospheric deposition of PCDD/PCDFs to plant tissues has been even less extensively studied. At Seveso, 2,3,7,8-T_4CDD analysis of grass in a fan-shaped area immediately SE of the factory was made at successive intervals of 1, 2 and 3 weeks following the July, 1976 accidental T_4CDD release (Homberger et al., 1979). These results are shown in Table 4.3.2.1C and confirm severe (up to 15,840 ng/g) contamination the week after the accident. The upper limit of contamination subsequently decreased to 35 and 23 ng/g during the second and third weeks, respectively, with 80 - 90% of the samples registering levels below 10 ng/g and 40% below the detection limit of 1 ng/g.

TABLE 4.3.2.1C

T_4CDD ANALYSIS OF GRASS SAMPLES COLLECTED IN A ZONE EXTENDING UP TO 900 m SE OF THE TCP FACTORY, SEVESO, ITALY

(Homberger *et al.*, 1979)

SAMPLING INTERVAL (Days Following Accident)	T_4CDD RANGE (ng/g)	% OF TOTAL SAMPLES IN RANGE
7-14	1300-15840	25
	100- 970	34
	21- 90	25
	2- 7	11
	1	5
15-21	11- 35	10
	1- 7.5	48
	1	42
22-28	10- 23	10
	1- 4	50
	1	40

In another study (Jansson and Sundstrom, 1982) involving environmental monitoring of contaminant deposition to vegetative surfaces no M_1-, D_2-, T_3-, or T_4CDFs were detected in pine needles collected in the vicinity (10 m) of a capacitor fire at a transformer station near Stockholm, Sweden. The detection limits for these analyses were not reported.

The former widespread use of T_4CDD-contaminated 2,4,5-T also has been evaluated as a source of contaminant deposition to forest and rangeland vegetation. Assuming a contamination level of 0.1 ug/g T_4CDD in 2,4,5-T applied at 2 pounds per acre, Kenaga and Norris (1983) estimated an initial T_4CDD concentration in sprayed grass at 23 pg/g.

The review by Norris (1981) suggested that maximum 2,4,5-T residues in forest vegetation can range up to about 300 ug/g for application rates up to 2.24 kg/ha active ingredient. Again, assuming a contaminant concentration of 0.1 ug/g T_4CDD this would equate to about 30 pg/g T_4CDD in exposed vegetation immediately after spraying. Actual measurements of vegetation sprayed with 2,4,5-T (Jensen et al., 1983; Sundstrom et al., 1979) at different rates and with different degrees of 2,4,5-T contamination provide fairly good agreement with the above estimates. These studies further indicate that the T_4CDD residue is very quickly dissipated with time, and as a result, vegetation contamination with T_4CDD is usually lower than expected based on the T_4CDD concentration in the formulation used.

4.3.2.2 Photochemical Synthesis

In addition to direct contamination from atmospheric emissions, accidental spills and the use of contaminated phenoxy and chlorophenol products, the possibility exists for photochemical generation of PCDDs and PCDFs. This topic has recently been thoroughly reviewed by Choudhry and Hutzinger (1982).

The ultra-violet component of sunlight is sufficiently energetic to liberate free radicals from both phenols and chlorinated aromatic compounds (Crosby et al., 1973). However, in the presence of adequate hydrogen donors dechlorination is the most likely reaction thereby decreasing the likelihood of PCDD formation from phenoxyacetic acids. The use of esterified phenoxy acids which would ensure the addition of hydrogen donors is suggested by Akermark (1978) as a practical method of reducing possible PCDD formation under field conditions.

Akermark (1978) points out that there has not been any evidence of PCDD formation in his studies or those of others (Crosby and Wong, 1973; Plimmer and Klingebiel, 1971), although he does indicate these negative findings may be related to the sensitivity of the analytical techniques at that time. The work of Sundstrom, et al. (1979) using 2,4,5-T under field conditions adds further support to these negative PCDD findings and should relieve any concerns that the large amounts of chlorinated phenoxyacetic acid derivatives presently spread through the environment may represent a potential source for the formation of PCDDs.

Other potential PCDD sources in nature are the various chlorinated phenols used as bactericides and fungicides. Irradiation of aqueous solutions of PCDD-free-sodium P_5CP have been shown to yield O_8CDD in small quantities (Crosby et al., 1973) while similar experiments with chlorinated biphenyls showed that traces of D_2CDF were detectable. P_5CP-treated wood exposed to artificial and natural sunlight also has been shown to give rise to H_6-, H_7- and O_8CDDs as photo products (Choudhry and Hutzinger, 1982).

Another photochemical process is the dechlorination of higher chlorinated PCDDs and PCDFs to lower chlorinated congeners. Although there is currently no evidence to support this concept in contaminated soils or vegetation in nature, the photolysis of O_8CDD and O_8CDF in solution has confirmed that chlorine atoms are removed preferably from the lateral positions on the carbon rings. As a result it is concluded that the most toxic PCDD isomer (2,3,7,8-T_4CDD) is not likely to be formed from the photolysis of higher PCDDs.

In the case of PCDFs even less is known concerning photoformation (Choudhry and Hutzinger 1982). The few artificial studies which have been conducted do, however, point to PCDF formation following the irradiation of some PCBs and polychlorinated diphenyl ethers.

In summation, it can be concluded that until positive analytical monitoring results from natural, ecosystem studies are published, the concept of photochemical formation of PCDDs and PCDFs in nature must be considered as a potential source, but of minor significance.

4.3.2.3 Degradation in Soils and Plants

Photodegradation

This process, which involves the cleavage of chemical bonds by light energy in the ultraviolet wave length range, has been thoroughly reviewed by Choudhry and Hutzinger (1982) Esposito et al. (1980) and by the NRCC (1981a). The following summary of the salient aspects of this process has been prepared from these reviews as well as from the earlier summaries by Helling et al. (1973), Crosby et al. (1973)and Crosby (1978). More recent information which addresses the use of photolytic degradation in remedial measures (desRosiers, 1983) and summarizes the role of this process in the disappearance of T_4CDD from a contaminated natural ecosystem also has been included (Young, 1983).

1. PCDDs and PCDFs readily undergo photolysis in the presence of hydrogen donors including alcohols, ethers, hydrocarbons, and natural products such as waxes.

2. Considering all the natural removal mechanisms and the fact that soil surfaces and plant foliage are the major direct recipients of pesticide sprays, photolysis appears to be the most significant degradation process; however, quantitatively little is known concerning the rate of photolysis in natural systems.

3. There are 3 major requirements for PCDD and PCDF photolysis:

   i. dissolution in a light-transmitting film;
   ii. presence of an organic hydrogen donor; and
   iii. ultra-violet light of appropriate wave length.

4. The primary photolytic degradation pathway involves dechlorination to less chlorinated congeners. The final end products of this sequence of reactions are unknown.

5. The higher chlorinated congeners such as O_8CDD are less reactive than the lower chlorinated compounds such as T_3CDD.

6. Impurities in common solvents can act as photolytic sensitizers and drastically improve the photodegradation process.

7. Because part of ultra-violet light reaches the earth's surface by reflection and dispersion from open sky, rather than from direct sunlight, full solar exposure is not required; however, as degradation is intensity-dependent, degradation in shade or under cloud cover would be slower.

From a practical aspect the application of T_4CDD to plant leaves via contaminated phenoxy herbicides results in the rapid breakdown of the T_4CDD upon exposure to sunlight. Photochemical degradation also occurs in soil, although at a slower rate, probably due to the penetration of the solution into the soil and the inability of light to penetrate to significant depths. Following the Seveso accident, a number of attempts were made to accelerate photodegradation through the addition of different solubilizing agents and hydrogen donors. In one study (Botre et al., 1978) a number of different cationic, anionic and nonionic surfactants were evaluated for their ability to solubilize 2,3,7,8-T_4CDD and to enhance photodecomposition using sunlight or artificial ultra-violet irradiation. The cationic surfactant 1-hexadecyl pyridinium chloride (HDPC) was found to be effective not only as a solvating agent but also in enhancing photodegradation. In another study in Seveso, olive oil in either 40% aqueous emulsion or

80% cyclohexanone solution was applied to a heavily contaminated area of grassland. After 9 days 80-90% of the T_4CDD was destroyed while controls remained virtually unchanged (Wipf _et al._, 1978).

Biodegradation

Although about 2 years have elapsed since the most recent reviews on microbial degradation (Esposito _et al._, 1980; NRCC, 1981a), very little additional quantitative information has been published.

Accordingly, this section will summarize and, where possible, update the information now known about the role of microbial degradation in the persistence of PCDD and PCDFs in soils.

1. In general, microbial degradation is the most ill-defined and poorly studied of all degradation processes.

2. The majority of work has centered on 2,3,7,8-T_4CDD with the only other information relating to 2,7-D_2CDD which is, at least, partially degraded in soils.

3. Current evidence indicates that PCDDs exhibit relatively strong resistance to biodegradation although they are not totally resistant.

4. Of 100 strains of microbes that had shown the ability to degrade persistent pesticides, only 5 showed limited ability to degrade 2,3,7,8-T_4CDD.

5. On the basis of the work performed at the Microbial Institute in Zurich, Switzerland, it has been concluded that microbes cannot contribute quickly or efficiently to the decontamination of 2,3,7,8-T_4CDD in soil although slow degradation might take place.

6. Under controlled laboratory conditions using carbon labelling techniques and nutrient-enriched, moist soils, T_4CDD has not been found to be metabolized in any significant manner; after 1 year only 1 - 2% of a hydroxylated metabolite was found and extractability of the T_4CDD decreased with long-term incubation.

7. In view of recent information concerning the difficulty of extracting PCDDs from soils (as a function of time), it can be concluded that much of the earlier work which did demonstrate limited biodegradability of PCDDs in soil cannot be taken as conclusive evidence for this degradative process.

As a result of their resistance to microbial activity, PCDDs are considered to be highly persistent with a reported half-life in soils in excess of 10 years (Kimbrough et al., 1984). From the work in Seveso (Di Domenico et al., 1980b), these estimates of longevity also appear to vary with time as after 1 month T_4CDD half-life was estimated at 10-14 months while after 17 months this value had increased to over 10 years. However, the more recent information concerning the extraction of T_4CDD from soils (Young and Arnold, 1983) also casts considerable doubt on these persistence estimates as new, more rigorous extraction techniques have been shown to recover T_4CDD which was formerly concluded to have been lost from test soils.

In the most recent report on microbial degradation, Matsumura _et al._ (1983a, b) demonstrated that after 4 months approximately 44% of the originally added T_4CDD in a model terrestrial ecosystem had been metabolized. This study was followed by another in which isolates of _Bacillus megaterium_ and _Nocardiopsis_ sp. were found to be able to degrade T_4CDD. The most important factor promoting this activity was the nature of the solvent used in applying the T_4CDD. Ethyl acetate gave the best results compared to dimethylsulphoxide or corn oil.

Summary

In summation, T_4CDD has been found to be a relatively stable compound capable of resisting microbial metabolism and thus can be expected to remain in the soil for long periods of time. However, on the basi of recent controlled model ecosystem studies, using carbon labelling techniques, there is evidence for very slow metabolism by some micro-organisms. Unfortunately, no information has been generated for any of the other PCDD or PCDF congeners.

4.3.2.4 Physical Transport in Soils

Vertical Mobility

The study of vertical mobility of PCDDs in soil over time is complicated by the effect of other factors which influence its fate in soil systems, i.e. microbial degradation, photodegradation, and volatilization. Thus, although several studies have been conducted which address the downward movement through the soil profile, there is as yet no general consensus on the mechanism involved.

In 1973, Kearney et al. found that the limited mobility of both 2,3,7,8-T_4CDD and 2,7-D_2CDD in soil was further decreased with increasing organic matter content in the 5 soil types studied. Matsumura and Benezet (1973) confirmed the very slow mobility of 2,3,7,8-T_4CDD. In a later study, Nash and Beal (1978), using both "microagroecosystems" and field plots, demonstrated that about 80% of applied 2,3,7,8-T_4CDD remained in the surface 2 cm of soil with trace levels being detected at depths of 8-15 cm.

The 1976 accidental release of 2,3,7,8-T_4CDD at Seveso, and the U.S. Air Force testing of Agent Orange herbicide have also provided unique opportunities to assess vertical movement under more natural conditions. Early studies at Seveso revealed that the T_4CDD migrated to a depth of 10 - 12 inches (Esposito et al., 1980); however, in a later study, Bolton (1978) found traces of contamination at a depth of over 30 inches. In another study, designed specifically to address this concept, Di Domenico et al. (1980c) found that vertical distribution of T_4CDD dropped sharply down to the 8 cm depth regardless of sampling location or degree of contamination. From 8-24 cm depth T_4CDD levels were less than those in the top 8 cm by at least 1 order of magnitude. Further examination of the top 2 cm of soil revealed that the highest levels were not found in the topmost layer (0-0.5 cm) but often in the second (0.5-1.0 cm or third 1.0-1.5 cm) layers. Again, no definite conclusions could be drawn concerning the mechanism responsible for the movement.

In a similar study, Belli et al. (1982) developed a set of complex statistical functions to describe downward movement of T_4CDD in the contaminated Seveso Zone A, based on data generated in surveys 3 months, 1 year and just over 2 years following the 1976 accident. The results indicated that penetration occurred predominately in the top 20 cm and during the first 10-15 months, and that contamination (about 1% of surface levels) to a depth of 1.5 m could be verified. On the basis of their findings, the authors speculate that T_4CDD is not removed from the surface but is subject to vertical migration.

In the U.S. Air Force studies, some vertical movement of 2,3,7,8-T_4CDD also was demonstrated (Young, 1974) at the Eglin A.F. Base, with the T_4CDD residues being confined to the upper 6 in. soil layer. These results have been summarized by Young (1983) and have been shown in Table 4.2.3.1A.

Volatilization

Very few studies have been conducted to assess or quantify the role of volatilization in the disappearance of PCDDs from soils. The summaries by Young (1983) and Esposito et al. (1980) have drawn together the limited work by Nash and Beal (1978) and Ward and Matsumura (1978), with the following conclusions:

1. In the "micro-agro-ecosystem" study approximately 10% of the T_4CDD applied as granular and emulsifiable silvex formulations was detected in air above the soil, confirming that T_4CDD has a low vapor pressure and that loss due to volatilization is limited.

2. Water-mediated evaporation of T_4CDD also may take place.

On the basis of these limited findings, Young (1983) concluded that volatilization and photodegradation were probably major routes in the disappearance of T_4CDD during and immediately after herbicide application to the Air Force test grids. He also suggested that water-mediated evaporation up through the contaminated soil surface also may have been a factor in the gradual disappearance of T_4CDD from this site.

In an effort to quantify the losses attributed to volatilization and other physical transport factors from a herbicide production facility that practiced on-site disposal in Jacksonville, Arkansas, Thibodeaux (1983) developed a transport model. The equation assumed that the boundary layer controlled the evaporation rate and that the chemical applied to the soil surface exerted its pure-component vapor pressure. Using a wind speed of 5 knots, a temperature of 25°C and a vapor pressure value of approximately 1×10^{-6} mm Hg for 2,3,7,8-T_4CDD, volatilization losses for a 753 m^2 "blow-out" area contaminated with T_4CDD at an average soil level of 1,300 pg/g were in the range of 120-1,200 g per year.

Wind and Water Movement

In a summary of the U.S. Air Force test results for the Eglin Base which received massive amounts of Agent Orange during the period of 1962-1970, Young (1983) concluded that the apparent vertical migration of 2,3,7,8-T_4CDD within the top 15 cm may not be due to leaching downward through the soil profile, but actually to the deposition of T_4CDD through the action of lateral wind and water movement. This was based on the presence within the 15 cm layer of discrete soil layers which differed from the underlying parent material and this in turn led to the conclusion that prevailing winds probably resulted in soil moving not only back and forth across Grid 1, but also between Grids 1 and 2. Water also moved contaminated particles from Grid 2 into the low-lying areas of Grid 1. Similar observations of the soils and T_4CDD contamination have been made by Di Domenico _et al._, (1982) for the contaminated Zone A at Seveso, Italy.

In another Seveso study (Monteriolo, 1982), hi-volume and dustfall air samplers were established and confirmed the presence of T_4CDD contaminated dust particles in the air. In the case of the small suspended particles, T_4CDD concentrations ranged from 0.17 - 0.43 pg/m^3 while the particles were reported as fallout at the rate of <0.1 - 0.6 $ng/T_4CDD/m^2/day$.

In his transport model, Thibodeaux (1983) estimated that soil entrainment losses from a 753 m^2 2,3,7,8-T_4CDD contaminated "blow-out" area with an average soil concentration of 1,300 pg/g T_4CDD, were 28 - 37 g/yr. This was based on an estimated deposition velocity of 2 cm/sec. This range is several times less than the corresponding loss figure developed for 2,3,7,8-T_4CDD volatilization.

4.3.3. AQUATIC FATE AND PERSISTENCE

PCDDs and PCDFs, as they are found in the environment, are subject to a variety of influences that result in their eventual destruction. In the aquatic environment, these influences can be divided into two major groups: non-biological and biological. The biological processes include microbiological decomposition, and to a lesser extent, metabolism within other aquatic organisms. The non-biological processes include transformation of these substances and direct chemical breakdown. Various authors have devised a variety of experiments to measure various portions of these biological and non-biological processes.

4.3.3.1 Non-biological processes

The non-biological processes controlling the aquatic concentration of PCDDs and PCDFs can be divided into two classes. The first class leads to direct breakdown of the contaminant and the second leads to translocation of the contaminant. The major physical breakdown process is photolysis. The translocational processes would be volatilization and possibly co-distillation and sorption to sediments and subsequent sediment transport.

Various authors have investigated the photolytic reactivity of PCDDs and PCDFs (Hutzinger et al., 1973; Mazer, 1982; Crosby, 1976). In the majority of these studies the PCDD was exposed to light in carefully prepared solvents (cf Section 4.3.2.3). Thus, the applicability of these studies to actual environmental conditions may be open to question. There have also been laboratory and field studies to examine the disappearance of T_4CDD from natural surfaces. These studies show that T_4CDD does indeed disappear from exposed natural surfaces. The studies did not reveal whether the disappearance was a photolytic destruction or some type of evaporation.

Volatilization

The half-life for volatilization of 2,3,7,8-T_4CDD is estimated to be 5.5 and 12 years from a pond and a lake, respectively (EPA Health Assessment Document, 1983). Generally, the greater the chlorination the lower the volatility and the longer the half-life.

By contrast, NRCC (1981a) estimate the half-life of 2,3,7,8-T_4CDD to range from 2 days in a 1 dm deep pond to 200 days in a 1 m deep pool. No experimental verification is available to substantiate these estimates.

Corbet et al., (1982), measuring 1,3,6,8-T_4CDD in air above treated ponds, found increasing amounts up to 1 day after treatment and afterwards decreasing amounts up to 4 days after treatment, and then a low steady rate of release. Volatization is not likely to be a major route through which 2,3,7,8-T_4CDD is removed from water; however, it may be for 1,3,6,8-T_4CDD.

Photolysis

The half-life for 2,3,7,8-T_4CDD as a result of photolysis is measured in terms of hours. Generally, the M_1-, D_2-, and T_3CDDs are more subject to photolysis and are therefore less persistent. The P_5-, H_6-, H_7-, and O_8CDDs are more resistant to photolysis and are therefore more persistent. However, H_7- and O_8CDD congeners are less active biologically (cf Section 3.6).

In theory, the PCDDs and PCDFs are subject to oxidation and hydrolysis; however, there is little published information on this specific topic.

Translocation

The physical translocation of TCDD has been observed by Ward and Matsumura (1978) to be a water borne process. Soil containing 2,3,7,8-T_4CDD may be subjected to wind and water erosion with subsequent reposition in a water body. Moreover, the evaporation processes observed by Ward and Matsumura would logically subject the 2,3,7,8-T_4CDD to greater levels of ultraviolet irradiation and photolytic degradation.

4.3.3.2 Biological Processes

Environmental Studies

Environmental studies of PCDD and PCDF can be separated into three categories, that is; laboratory model ecosystems, pond studies and studies on known areas of contamination. In the laboratory model ecosystem, usually 2,3,7,8-T_4CDD is added to an aquarium containing elements of an aquatic ecosystem. In this relatively well controlled environment, the movement of the contaminant is monitored over a period of time. In the pond studies, the contaminant is added to an outdoor pond and again the movement and decomposition is monitored over a period of time. The in situ studies involve areas that have had a known history of dioxin contamination. In this situation, various aspects of the contaminated environment are monitored over time to document the movement and degradation of the substances.

Model Ecosystem Studies

Yochim et al., 1978; Isensee and Jones, 1975; and Matsumura et al., 1983a, 1983b have run model ecosystem studies to examine the movement, deposition, and persistence of PCDD and PCDF in a controlled environment. Such studies are based on the assumption that nothing will be limiting to the free functioning of that closed system. These studies were designed essentially to establish a mass balance of the contaminant in the model.

Matsumura et al. (1981), using an outdoor pond, estimated the half-life of 2,3,7,8-T_4CDD to be

1 year. In the outdoor pond, fathead minnows accumulated 2,3,7,8-T_4CDD to a peak after 10 days and then declined to a steady state after 40 days. After 10 days, 2,3,7,8-T_4CDD was not detected. Sediment levels of 2,3,7,8-T_4CDD peaked at about 5 days and them slowly declined. Concentrations of 2,3,7,8-T_4CDD in pond weeds (Elodea nuttali and Ceratophyllum demersum) peaked at 5 days and continued to decline to steady state at 50 days. After 1 year, 49.7% of the applied 2,3,7,8-T_4CDD remained unmetabolized. The authors consider evaporation, photochemical degradation and microbial degradation as the most important processes to remove T_4CDD from the aquatic environment.

Yochim (1978), using a model ecosystem and radioactive 2,3,7,8-T_4CDD, found bioaccumulation constants for a variety of aquatic organisms to be between 2,000 to 6,000. The author added the comment that these values are similar to those of Isensee and Jones, 1975.

Isensee (1978) established a correlation coefficient of 0.94 or more between the concentration of 2,3,7,8-T_4CDD in water and organisms. The author suggests that the amount of 2,3,7,8-T_4CDD accumulated by the organisms is controlled almost entirely by the amount available in the water. A large proportion of dioxin added to the soil/sediment stays in the sediment.

Isensee (1975) found the amount of 2,3,7,8-T_4CDD accumulated by aquatic organisms is controlled almost entirely by the amount of T_4CDD available in the water, as opposed to total ecosystem exposure.

The author added radioactive 2,3,7,8-T_4CDD to soil and then flooded it. Fish and algae were added to the water. It was found that 85-99% of the 2,3,7,8-T_4CDD remained in the soil but there was still some accumulation in all organisms. After the treated soil, water and algae had the next highest amounts, while other components of the biomass contained substantially less. In a comparative examination of the work of several other authors, Isensee and Jones (1978) state that 2,3,7,8-T_4CDD had about 20% (1/5) the bioaccumulation potential of DDT and aldrin. They also found a strong correlation between the water concentration and the concentration in various test organisms, thus demonstrating that the water-borne contaminant is important in controlling the organism concentration of 2,3,7,8-T_4CDD.

Environmental Analyses

In contrast to the laboratory studies, Czuczwa and Hites (1984) examined various sediment cores from Lake Huron for evidence of PCDD and PCDF contamination. The cores were collected from the Saginaw River, Saginaw Bay, and the southeast quadrant of Lake Huron (offshore from Grand Bend and Kincardine). All of these sediment cores showed congener distribution profiles. Five different series of isomer groups of the PCDFs and PCDD were examined. Of the PCDFs and PCDDs, H_7CDF and O_8CDD consistently predominated. Consistently, T_4CDF and T_4CDD were both among the lowest concentrations in the profile. Concentrations of the PCDFs and PCDDs in sediments ranged from 40 ng/g in Saginaw River to 0.3 ng/g in Lake Huron.

Ward and Matsumura (1978) incubated 2,3,7,8-T_4CDD in lake water with and without lake sediment. Most of the 2,3,7,8- T_4CDD accumulated in the sediment. In the water/ sediment incubation system, they estimated half-life to be 600 days. In water alone, it was longer but not stated (in excess of 700 days). They found that most of the T_4CDD was accounted for; thus, there was not much effect of light as the 2,3,7,8-T_4CDD was in the sediment.

From these studies, a general conclusion can be drawn that PCDDs and PCDFs are more frequently found at measurable concentrations in sediment than in water. The work of Ward and Matsumura (1978) and Matsumura et al. (1981) suggests that PCDDs and PCDFs in the sediment can be expected to have a half-life of 1 - 2 years. However, the evidence of Czuczwa and Hites (1984) indicates that PCDDs and PCDFs may persist in lake sediments for decades.

4.3.4 BIOLOGICAL ACCUMULATION/MAGNIFICATION

4.3.4.1 Wildlife and Domestic Animals

The accumulation and biomagnification of PCDDs by wild and domestic animals has been thoroughly reviewed by Esposito et al. (1980) and Norris (1981). In view of the large number of studies which were described in each, no attempt will be made here to review these summaries in detail. Rather, the major conclusions which were reached have been summarized. Toxicological aspects of these studies are covered in Section 3.5.1.2

Norris (1981) concludes that the various test results indicate that if 2,3,7,8-T_4CDD is present in the environment in a form which is available to organisms, then it will be accumulated by them if exposed. This accumulation will vary depending on the physico-chemical properties of the compound as well as its persistence and availability in the environment. Monitoring results indicate that "substantial bioaccumulation of T_4CDD (sufficient to produce residue levels in excess of 10 ng/kg 2,3,7,8-T_4CDD in the majority of the [animal] population) is not occurring in animals in or near areas treated with 2,4,5-T or Silvex in operational programs". Norris attributes this to mechanisms of degradation and dilution which operate in the natural environment. In their review, Esposito *et al.* (1980) do not make any general statements concerning bioaccumulation or biomagnification. Thus, a brief summary of the conclusions reported in the various studies cited is warranted.

1. On the basis of the animal studies conducted at Seveso, it can be concluded that T_4CDDs do accumulate in environmentally exposed wildlife at average concentrations only slightly higher than the corresponding average surface soil results (Frigerio, 1978). Earthworms which ingested contaminated soil accumulated 2,3,7,8-T_4CDD by a 14.5 fold-factor on average (Martinucci *et al.*, 1983).

2. On the basis of studies with rodents and other animals on the U.S. Eglin Air Force Base, it was concluded that T_4CDDs do accumulate in animal tissues to about the same magnitude as the corresponding soil concentrations; however, in some cases T_4CDD bioaccumulation significantly exceeded the soil level suggesting a potential for biomagnification (Young, 1974).

3. In a study of milk from cows grazing on pasture which had received 2,4,5-T, no 2,3,7,8-T_4CDD was detected at a detection limit of 1 pg/g (Getzendaner et al., 1977).

4. In a similar study, milk samples were collected throughout the Seveso area immediately after the accident and revealed significant contamination (up to 7,919 ng/L) (Fanelli et al., 1980b).

5. In an insecticide-feeding study, no evidence of bio-accumulation was apparent in the fat of cattle over a 147 day period (Shadoff et al., 1977).

The most recent publication dealing with bioaccumulation and bioconcentration factors (BCF) is the review by Kenaga and Norris (1983). Their main conclusions with respect to T_4CDD are summarized below:

1. **Bioconcentration factors (biomagnification) for soil to mammals and food to mammals, are very low.**

2. **In the case of soil, the low BCFs are related to the low availability (high adsorption to soil) and concomitant low degree of exposure to T_4CDD.**

Reports on PCDD levels in domestic animals as related to human diet are limited and they have been well summarized in Table 6-1 of a recent report (NRCC, 1981a). Unfortunately, a large proportion of the data is taken from areas around Seveso, site of a major T_4CDD industrial accident, and from areas outside Canada. Thus they cannot all be considered realistically applicable in Ontario. North American data are reviewed below.

P_5CP in contaminated animal feed, in saw dust or shavings from treated wood used as animal bedding or in treated farm structures has been implicated in poisoning episodes in cattle, swine, chickens and domestic pets. More recently PCDD and PCDF contamination of P_5CP has been suggested as the cause of toxicity in animals exposed to materials containing high levels of P_5CP.

Livers of Ontario chickens raised on wood shaving litter containing measurable levels of P_5CP were discolored and contained detectable levels of PCDDs (Ryan and Pilon, 1982). Levels of PCDDs ranged from ND - 1.42 ng H_6CDDs/g; ND - 0.57 ng H_7CDD/g and ND - 0.66 ng O_8CDD/g. Levels of O_8CDF ranged from ND - 0.13 ng/g. Skin, liver, kidney and brain of young Ontario pigs, which died in farrowing pens freshly treated with P_5CP, contained 32 - 70 pg H_6CDD/g, 790 - 1670 pg H_7CDD/g and 27 - 4800 pg O_8CDD/g (Ryan, 1983). Deaths of these young pigs ceased when the original floor was covered with untreated plywood.

It should be pointed out that these residues were found on sick or dead animals. Recent and proposed changes in the use patterns of P_5CP in Canada described in Section 4.2.2.1 should substantially reduce exposure of domestic animals to P_5CP and its PCDD and PCDF contaminants.

At a recent meeting in Ottawa (Fourth International Symposium on Chlorinated Dioxins and Related Compounds, October 16 - 18, 1984), several speakers reported on the PCDD and PCDF content of some dietary items as follows:

1) Poultry - A low proportion (12%) of samples of Canadian chicken fat sampled in 1980 contained up to 400 pg/g of PCDDs and PCDFs. These are mainly H_6-, H_7 and O_8CDDs with O_8CDD forming about 60% of the total residues. No T_4- and P_5-CDDs and CDFs were found and O_8CDF was not detected.

2) Pork - Samples of Canadian pork fat can contain up to 500 pg/g PCDDs, mainly H_7- and O_8CDD. O_8CDD forms up to 85% of the total residue found.

3) Beef - 10% of North American slaughterhouse samples contained H_6-, H_7- and O_8CDD in the low pg/g range.

4) Eggs - Canadian hens eggs can contain up to 100 pg O_8CDD/g.

Clearly, diet can be a significant contributor to PCDD and PCDF intake by man. Given the range and concentrations of higher chlorinated PCDDs and PCDFs found in human adipose tissue (Section 4.3.4.5) and the restricted range of T_4- and P_5-CDDs and CDFs in freshwater fish (Section 4.3.4.4), other components of the diet should be analyzed to determine their contribution to dietary intake of PCDDs and PCDFs (e.g. market basket survey).

4.3.4.2 Vegetation

Information concerning the accumulation of PCDDs by terrestrial plants is important not only in terms of its impact on the removal and decontamination of affected soils, but also as a factor in the food web leading to possible biomagnification through ingestion by animals. Unfortunately, only a limited number of studies have been conducted to characterize and assess the magnitude of plant accumulation.

The early work in the 1970s has been thoroughly reviewed by Esposito et al. (1980) and Norris (1981) and can be summarized briefly as follows:

1. In an uptake study involving oats and soybeans, only a limited amount (0.15%) of T_4CDD and D_2CDD applied to the soil was subsequently detected in the above-ground plant portions.

2. In the same study, T_4CDD and D_2CDD applied in surfactant to the surface of plant leaves was not translocated from the treated leaves of oats or soybean to other plant parts.

3. A year after the accident at Seveso, concentrations of T_4CDD in leaves of fruit trees of species growing in the most severely contaminated Zone A were 3-5 times higher than in the fruit but lower than in twigs or cork tissue; these results indicated that T_4CDD was biologically transported from the soil by the plants.

4. Concentrations of T_4CDD in above and below ground portions of garden vegetables also growing in the contaminated Zone A at Seveso, revealed that the aerial portions contained lower levels than the roots/tubers; however, the levels were usually less than the concentrations in the soil.

5. In a controlled uptake experiment, it was shown that soybean plants accumulated 430 pg/g T_4CDD after 64 days in soil treated with Agent Orange herbicide containing 14 ug/g T_4CDD.

6. No T_4CDD was found in plant seeds from species growing in an area where tons of 2,4,5-T-containing herbicides had been applied in tests of spraying equipment by the U.S. Air Force.

In a more recent summary of the Seveso data, Monteriolo _et_ _al._ (1982) suggest that when TCDD soil contamination drops from 1,000 ug/m^2 (6 ng/g) to 100 ug/m^2 (0.6 ng/g) or even down to 1-2 ug/m^2 (0.006 ng/g), underground vegetables exhibit an even greater drop in T_4CDD contamination. A factor of 10 is suggested as the magnitude by which underground vegetables are lower than corresponding soil samples. This was confirmed in field investigations, where wheat and rye ears in Zone R were found to display less than 0.5 pg/g T_4CDD at soil levels of about 6 pg/g. Wipf _et_ _al._ (1982) also summarized the Seveso monitoring data and presented vegetation data (Table 4.3.4.2A) for samples collected in Zones B and R in the fall of 1977. These results (Table 4.3.4.2A) confirm the lack of any significant plant uptake of T_4CDD from marginally contaminated soils. In the most contaminated Zone A, Wipf et al. (1982) examined the partitioning of T_4CDD in apple, pear and peach fruit growing in soil containing a T_4CDD concentration of 10,000 pg/g. In no case was any of the fruit (whole) found to contain levels greater than 37 pg/g. Moreover, 95% of the T_4CDD (high of 137 pg/g) was found in the peel, suggesting contamination from surface dust rather than plant uptake.

In a summary of the Eglin Air Force Base studies, Young (1983) reported on a plant uptake study which was conducted in 1978 and again in 1979. The study involved isolating square plots by excavating trenches around them and then harvesting all aerial plant portions as well as the complete 0-5, 5-10 and 10-15 cm increments of soil. The roots in each soil layer were subsequently removed and pooled by species for analysis. The results are summarized in Table 4.3.4.2B. In view of the similar levels between roots and soil, it was suggested that a 'passive' process of uptake was responsible and that plants would account for 5.3×10^{-4} g T_4CDD removal from the Test Grid annually.

Summary

On the basis of these and other related studies, it has been concluded (Kenaga and Norris, 1983) that plants do not bioconcentrate T_4CDD from the soil.

TABLE 4.3.4.2A

T_4CDD CONCENTRATIONS IN VEGETATION GROWN CLOSE TO THE GROUND IN ZONES B AND R, SEVESO, ITALY, 1977*

Plant Material	T_4CDD Soil Concentration (pg/g)	T_4CDD in Plant Material (pg/g)
grass	150	2
silverbeet (leaves)	200	0.9
millet	2	0.9
sage (leaves)	36	5
cauliflower	10	1
cauliflower leaves	10	1
chicory	10	3.5
cabbage	10	0.7
forage plant	38	1.7
cucumber	15	0.4

* from Wipf *et al.*, 1982.

TABLE 4.3.4.2B

PLANT UPTAKE DATA FOR PERENNIAL GRASSES AND BROADLEAF PLANTS FROM STUDIES CONDUCTED ON SELECTED SITES OF GRID 1, EGLIN A.F.B.*

Sample Type	Plant Portion or Soil Depth (cm)	2,3,7,8-T_4CDD Concentration (pg/g)
Grasses	culm/leaves	10
	crown	270
	roots	710
Broadleaf plants	stem/leaves	75
	roots	760
Soil	0-5	95
	5-10	510
	10-15	630

*From Young (1983)

4.3.4.3 Aquatic Biota

Laboratory Studies

Assessments of the biological accumulation or magnification are carried out in a controlled exposure wherein the concentration of the contaminant in both the medium and the organism can be measured. Several authors have measured the accumulation of PCDDs and PCDFs in a variety of aquatic organisms Table 4.3.4.3A. In these studies the exposure times varied from at least 4 to 40 days. However, all the authors felt that an equilibrium condition had been achieved. The accumulation factors varied from 469 for 1,3,6,8-T_4CDD in rainbow trout to 7125 with *Daphnia magna* exposed to 2,3,7,8- T_4CDD (Table 4.3.4.3A). This range of bioaccumulation is moderate when compared to PCB (Mayer *et al.*, 1977) or DDT (Macek *et al.*, 1979,)

Similar selective uptake of 2,3,7,8-T_4CDD by fish from municipal fly ash was observed in a joint project by EPA and the Dow Chemical Company (study cited in Long and Hanson, 1983). Carp fingerlings were put into a fish tank with some municipal fly ash of which the 2,3,7,8-T_4CDD content was about 160 pg/g. That represented about 0.48% of the total T_4-isomers, all of which were present. After thirty days in the tank, the fish were analyzed and 84% of what was found in the fish was 2,3,7,8-T_4CDD. This selective accumulation in fish also may be a result of selective absorption and/or metabolism.

TABLE 4.3.4.3A

BIOLOGICAL ACCUMULATION OF TCDD IN AQUATIC ORGANISMS

ORGANISM	COMPOUND	EXPOSURE PERIOD (days)	ACCUM. FACTOR	AUTHOR
Oedogonium cardiacum	2,3,7,8-T_4CDD	7	2075	Isensee, 1975
"	"	7	2083	Yochim, 1978
Elodea nuttali	"	40	2000	Tsushimoto,1982
Ceratophyllum demersum	"	40	2000	"
Physa sp.	"	7	2095	Isensee, 1975
Helosoma sp.	"	7	3731	Yochim, 1978
Daphnia magna	"	7	7070	Isensee, 1975
"	"	7	7125	Yochim, 1978
Gambusia affinis	"	7	4075	Yochim, 1978
Pimephales promelas	"	40	2500	Tsushimoto,1982
"	1,3,6,8-TCDD	>4	1061	Corbet, 1983
Salmo gardneri	"	>4	469	Corbet, 1983

4.3.4.3 Field Studies

Fish from Lake Ontario were analyzed for 22 T_4CDD isomers and compared with control fish from Lake Erie and the eastern Pacific Ocean. (Ryan _et_ _al._, 1983b). Presumptive evidence of only 2,3,7,8-T_4CDD was found in Lake Ontario fish. The concentration was positively related to both the fat content and size of fish. Neither of the control samples showed any 2,3,7,8-T_4CDD. In addition, the Pacific coast samples had no detectable 2,3,7,8-T_4CDF. The authors concluded by suggesting that the T_4CDD and T_4CDF contamination in the fish is related to chemical manufacturing and subsequent loss of waste materials into the Lake Ontario basin. Combustion and atmospheric deposition were considered less likely sources.

Stalling _et_ _al._ (1983a) examined fish from all the Great Lakes for evidence of PCDD. Fish from Lake Superior and Lake Siskiwit contained no detectable PCDD. However, fish from Lakes Michigan and Ontario contained 2,3,7,8-T_4CDD at 5 and 33 pg/g, respectively. No other dioxin isomers or congeners were detected.

Stalling _et_ _al._ (1983b) also examined aquatic sediments and fish living over sediments for evidence of PCDF contamination. Very little 2,3,7,8- T_4CDF was found in the sediment but larger amounts were found in the fish; conversely, very little O_8CDF was found in the fish but larger amounts were in the sediment. The authors suggest that there is apparently selective, preferential uptake and/or inability to excrete and/or metabolize the T_4CDF.

Petty et al. (1983) found similar results in examining sediment and yellow perch from Woods Pond, Massachusetts. (Table 4.3.4.3B)

TABLE 4.3.4.3B

PCDF DISTRIBUTION IN A POND KNOWN TO BE CONTAMINATED BY AROCLOR 1260

	PCDF CONCENTRATION (ug/kg)		
	Sediment	Yellow Perch	Aroclor 1260
T_4CDF	0.005	1.06	290
P_5CDF	0.009	0.64	1330
H_6CDF	0.15	0.44	1810
H_7CDF	0.92	0.005	780
O_8CDF	0.27	0.005	29
TOTAL PCDF	1.35	2.14	4240
PCB (ug/g)	60	170	Neat
PCDF/PCB Ratio	22.5×10^{-6}	12.6×10^{-6}	4.24×10^{-6}

Petty et al., (1983)

Stalling et al. (1983b), in a later paper, went on to state that isomer distribution of PCDD and PCDF in sediments appear to differ markedly from those measured in the surrounding biota and factors controlling isomer distribution in the environment are largely unknown. Evidence for preferential accumulation of T_4CDD, T_4CDF and P_5CDF in aquatic biota is presented.

For organisms in water, it has been suggested that PCDD and PCDF bioaccumulate moderately due to their hydrophobic nature. PCDDs and PCDFs are tightly bound to soil and sediment or carbonaceous material (Kenaga and Norris, 1983).

4.3.4.4 Residue Levels in Fish From Ontario Waters

Isomer-specific determination of 2,3,7,8-T_4CDD in fish from Ontario waters has been performed by both MOE and the New York State Health Department (NRCC, 1981a).

The M.O.E. Dioxin Laboratory has also analyzed many fish samples from the Great Lakes, interconnecting waterways and some tributary streams. Except for trace amounts of 2,3,7,8-T_4CDD in lake trout from Peninsula Harbour, the fish of Lakes Superior, Huron and Erie appear to be free of this contaminant (Table 4.3.4.4A).

In contrast, 2,3,7,8-T_4CDD is often detected in samples from the Niagara River and Lake Ontario. M.O.E.'s New Releases (1981, 1982) show that 2,3,7,8-T_4CDD levels are highest in young-of-the year spot-tailed shiners (less than one year old) collected from the mouth of Cayuga Creek, and larger lake trout from the western end of Lake Ontario. Cayuga Creek receives drainage from the Love Canal area and has been shown to contain 2,3,7,8-T_4CDD (Smith et al., 1983b). Detectable residues of 2,3,7,8-T_4CDD residues in spot-tail shiners less than one year old indicates the usefulness of this fish as biological monitors to identify point sources of PCDD/PCDF contamination.

In addition, federal government tests of a small number of American eel and smelt taken from the eastern part of Lake Ontario indicate the occasional occurrence of T_4CDD levels above 20 pg/g.

Lake trout longer than 45 cm (18 inches) from western Lake Ontario are restricted from human consumption as levels of 2,3,7,8- T_4CDD are likely to exceed 20 ppt (Ontario Sportfish Consumption Guideline). This is the only case in Ontario where sportfish consumption is restricted due to 2,3,7,8-T_4CDD contamination.

Table 4.3.4.4B gives an estimate of the number of fish taken from Lake Ontario by anglers. Of the grand total taken, 39,000 are in the trout class (lake, brown and rainbow trout).

A survey by M.O.E. of fish consumption practices by anglers showed that 85% of the respondents fell into a category where fish consumption ranged from one meal a year to one meal a week. Of the people in this category, 80% would eat a mean portion size of 300 g (range 200 to 450 g).

Recent data (MOE, unpublished), studies discussed in sections 4.2.4.3. and recent reports (Fourth International Symposium on Chlorinated Dioxins and Related Compounds, Ottawa, October 16-18, 1984) indicate that some Great Lakes fish which do not contain 2,3,7,8-T_4CDD may contain other 2,3,7,8-substituted T_4- and P_5- CDDs and CDFs as well as 2,3,7,8-T_4CDF. 2,3,4,7,8-P_5CDF has been found in the 8 - 15 pg/g range in some of these fish. Higher chlorinated PCDD or PCDF congeners are not generally detected. Consequently, freshwater fish are apparently not a source of higher chlorinated PCDDs and PCDFs to man.

These area-specific data indicate further monitoring of PCDD and PCDF levels in Ontario fish is required. Future sport fish consumption guidelines should be updated in light of these data.

TABLE 4.3.4.4A

2,3,7,8-T_4CDD IDENTIFIED IN GREAT LAKES FISH

LOCATION	SPECIES	NUMBER SAMPLED	LENGTH (cm)		2,3,7,8-T_4CDD (pg/g)	
			MEAN	RANGE	MEAN	RANGE
Lake Superior						
Thunder Bay	Lake Trout	6	-	-		ND @ 10 pg/g
Peninsula Harbour	Lake Trout	11	44.9	34.0 - 64.0	0.45	ND - 2.0
Lake Huron						
Point Edward	Lake Trout	6	-	-	-	ND @ 10 pg/g
Owen Sound	Rainbow Trout	6	-	-	-	ND @ 10 pg/g
St. Joseph Island	Walleye	6	-	-	-	ND @ 10 pg/g
Detroit River						
Fighting Island	Yellow Perch	3	-	-	-	ND @ 10 pg/g
Lake Erie						
Port Dover	Rainbow Trout	3	-	-	-	ND @ 10 pg/g
Nanticoke	Spottail Shiner	-	-	-	-	ND @ 1 pg/g
Grand River:						
- Canagague Cr. Upstream of Elmira	Rock Bass	2	-	-	-	ND @ 10 pg/g
- Downstream of Elmira	White Sucker	4	-	-	-	ND @ 10 ppt
	Rock Bass	1	-	-	-	ND @ 10 pg/g
	Brown Bullhead	1	-	-	-	ND @ 10 pg/g
Grand River @ Breslau	Smallmouth Bass	8	24.8	18.0 - 35.0	ND	ND @ 1 pg/g
	Rock Bass	5	16.8	14.0 - 19.0		

TABLE 4.3.4.4.A

2,3,7,8-T_4CDD IDENTIFIED IN GREAT LAKES FISH

LOCATION	SPECIES	NUMBER SAMPLED	LENGTH (cm)		2,3,7,8-T_4CDD (pg/g)	
			MEAN	RANGE	MEAN	RANGE
Niagara River						
Upper Niagara R.	Yellow Perch	6	-	-	-	ND @ 10 pg/g
Miller Cr.	White Sucker	1	-	-	-	ND @ 10 pg/g
	Smallmouth Bass	1	-	-	-	ND @ 10 pg/g
Fort Erie (Ontario) Frenchman's Creek	Spottail Shiners	1	4.9±0.3	-	15	(1981)
		1	-	-	N.D.	(1983) -
Opposite 102 nd St. Disposal Area, New York	Spottail Shiners	2	4.9±0.3	-	7.5	4, 11
Mouth of Cayuga Ck., New York	Spottail Shiners	2	-	-	59	58, 60
Petite Flume, North Tonawanda	Spottail Shiners	1	-	-	120	-
Lower Niagara R.						
Lewiston, N.Y.	Spottail Shiners	-	-	-	-	ND @ 1 pg/g
Peggy's Eddy, N.Y.	Spottail Shiners	2	5.3±0.4	-	7	3, 11
Queenston, Ontario	Spottail Shiners	-	-	-	-	ND @ 1 pg/g
	American Eel	5	-	-	-	ND @ 10 pg/g
	Walleye	1	-	-	-	ND @ 10 pg/g
	Rainbow Trout	1	-	-	-	ND @ 10 pg/g
	Northern Pike	1	-	-	-	ND @ 10 pg/g
	Muskellunge	1	-	-	-	ND @ 10 pg/g
	Yellow Perch	6	-	-	-	ND @ 10 pg/g
Niagara-on-the-Lake, Ontario	Spottail Shiners	2	5.5±0.3	-	13.5	13, 14

Page 3 of 3

TABLE 4.3.4.4 A

2,3,7,8-T_4CDD IDENTIFIED IN GREAT LAKES FISH

LOCATION	SPECIES	NUMBER SAMPLED	LENGTH (cm)		2,3,7,8-T_4CDD (pg/g)	
			MEAN	RANGE	MEAN	RANGE
Lake Ontario						
Jorden Harbour	Brown Trout	13	-	-	-	7 of 13 positive: ND** to 19 pg/g
	White Bass	6	-	-	-	2 of 6 positive: ND** to 19 pg/g
Clarkson	Lake Trout	11	60	50.9 - 69.5	27.4	17.0 - 57.0
Credit River	Coho Salmon	8	-	-	-	ND @ 10 pg/g
Humber Bay	Lake Trout	6	47.3	43.0 - 51.0	7.8	6.0 - 8.0
	Rainbow Trout	3(c)	17.7	15.0 - 21.0	8.0	7.0 - 10.0
Toronto-Hearn Generating Station	Rainbow Smelt	8	-	-	-	1 positive: ND** to 11 pg/g
Bluffers Park - Scarborough	Lake Trout	5	-	-	-	3 positive: ND** to 1 pg/g
Ganaraska R. @ Port Hope	Rainbow Trout	7	62.0	45.8 - 73.9	8.3	7.0 - 11.0
Bay of Quinte	Yellow Perch	6	-	-	-	ND**
	White Perch	6	-	-	-	1 positive: ND** to 16 pg/g

ND** = Not detected at 10 pg/g.
ND = Not detected. (detection limit not specified)
- = Data not recorded.

TABLE 4.3.4.4B

1980 SURVEY OF ONTARIO RESIDENT SPORT FISHERMEN NUMBERS (1000'S) OF FISH CAUGHT AND KEPT BY ADULTS ON SELECTED WATER BODIES

Lake Superior	Caught	Kept
Panfish, total	65	52
(Smelt	294	234)
Trout, total	93	88
TOTAL, all species	486	401
Niagara River		
Bass, total	75	8
Panfish, total	339	148
(Smelt	832	735)
TOTAL, all species	1358	917
Lake Ontario		
Bass, smallmouth	54	28
Bass, total	199	101
Coarse fish	197	34
Northern Pike	79	41
Panfish	544	247
Perch	267	140
Salmon, coho	126	79
Salmon, total	182	110
Smelt	4622	4156
Trout, total	67	39
Walleye	136	107
TOTAL, all species	6119	4846

NOTE: Estimates for all species/lake combinations are based on a minimum of 25 reports from question 3 where the catch (not the harvest) exceeded zero. Estimates not meeting this minimum requirement for sample size are considered unreliable, and are not displayed, except when they contribute significantly to a total and are bracketted.

4.3.4.5 **Residue Levels in Humans**

Human exposure to PCDDs and PCDFs may occur occupationally, accidentally, or from dietary intake and/or environmental exposure eg. air, water or soil.

Levels of PCDDs or PCDFs in human tissues following occupational exposure to contaminated commercial products such as chlorophenol herbicides or wood preservatives or PCBs has been monitored only in a few cases. Rappe et al. (1983c) analyzed blood samples from workers in a textile plant, a sawmill and a tannery in Sweden. The workers had been exposed to P_5CP or 2,3,4,6- T_4CP and their derivatives. PCDD and PCDF congeners were found in all samples in concentrations (<1 to 304 pg/g) which could be related to the nature of the exposure, the duration of exposure and the isomers present in the commercial formulation. Unfortunately background samples from the general population were not obtained during these studies.

Accidental exposure to PCDFs following ingestion of PCB-contaminated rice oil affected thousands of people in Japan in 1968 and in Taiwan in 1979. The resulting clinical syndrome called Yusho in Japan and, Yu-cheng in Taiwan was characterized by chloracne, increased eye discharge, swelling of the upper eyelids, and pigmentation of the face, eyelids, nails, and gums (Masuda and Yoshimura, 1984).

In the case of the Yusho incident, the rice oil contained 5 ug total PCDFs/g and the adipose and liver tissues of deceased patients contained 6 to 13 ng PCDF/g and 3 to 25 ng PCDF/g, respectively. No PCDFs (detection limit - 50 pg/g) were found in the adipose tissue or liver of traffic accident victims from the general population (Kunita et al., 1984).

PCDF composition of these patients was much less complex than the PCDF mixture ingested. Forty PCDF congeners were identified in the Yusho rice oil, most of which have not been found in the Yusho patients. For example, 2,3,6,7-T_4CDF, 2,3,4,6,7-P_5CDF, 1,2,6,7,8-P_5CDF, 1,2,3,4,8-P_5CDF and 1,2,3,4,6,7-H_6CDF all present in the rice oil, have not been found. On the other hand, 2,3,6,8-T_4CDF, 2,3,7,8- T_4CDF, 1,2,4,7,8-P_5CDF, 1,2,3,7,8-P_5CDF, 2,3,4,7,8- P_5CDF and 1,2,3,4,7,8-H_6CDF were retained by the Yusho patients (Masuda and Yoshimura, 1984).

Rappe et al. (1979c) pointed out that none of the PCDFs detected in human liver approximately 1 year after exposure had adjacent, unsubstituted carbon atoms. 2,3,4,7,8-P_5CDF was the predominant PCDF congener retained even 9 years after the original exposure (Kuroki and Masuda, 1978). The PCDF congeners that persisted were the potentially toxic or biologically active 2,3,7,8- substituted ones which also appear to be more resistant to metabolic detoxification.

Rappe (1984) reported that all of the 40 normal human blood samples which he has analyzed were negative for 2,3,4,7,8-P_5CDF at the 10 pg/Kg level of detection. Rat studies (Yoshihara et al., 1981) and PCDF residues levels in deceased Yusho patients

(Kuroki and Masuda, 1978) indicate that 2,3,4,7,8-P_5CDF is the most readily bioconcentrated and most persistent PCDF congener.

PCDFs were not detected (detection limit - 10 pg/g) in the blood of Japanese workers handling fresh or used PCB formulations (Kunita et al., 1984). Similarly, no PCDFs were detected in the blood of healthy Japanese not exposed to PCBs or contaminated rice oil.

As discussed in Section 3.2.7 2,3,7,8-T_4CDD accumulates preferentially in the adipose tissue and liver of laboratory animals and monkeys. Thus, an analysis of human fat samples for 2,3,7,8-T_4CDD may provide a way to estimate prior human exposure to PCDD's. Three separate groups of researchers presented such data recently at an American Chemical Society meeting (ACS Meeting, Washington, 1983) which are summarized in Table 4.3.4.5A. One group of data are from Ontario.

TABLE 4.3.4.5A

TCDD LEVELS IN HUMAN ADIPOSE TISSUE

REFERENCE	SUBJECTS	NUMBER OF SAMPLES	NUMBER AT or ABOVE DETECTION LIMIT	TCDD CONCENTRATION RANGE (ppt) (Outlier in brackets)	MEAN TCDD CONCENTRATION
(Ryan, and Williams 1983)	People older[a] than 50, all of whom died of natural causes while hospital-ized in Ontario	22	22	4.1-21.8 (130)[b]	12.4 ± 5.8 ppt (Kingston) 8.6 ± 4.4 ppt (Ottawa)
(Stanley, *et al.* 1983)	U.S. urban population	6	6	5-12	-
(Hobson, *et al.* 1983)	10 U.S. veterans with no Vietnam Service[c]	10	6	7-14	5.7 ± 3.1 (for 10 non-Vietnam veterans)

[a] Adipose tissue degeneration commences upon death. Since these results are from cadavers, the dioxin levels may be elevated as compared to samples from living persons.

[b] Explanation for high value (i.e. outlier) is not known.

[c] To determine earlier exposure to Agent Orange which was applied in Vietnam. Results of the controls ("unexposed") only are quoted here.

Detailed discussion of the results can be found in the references themselves. The mean 2,3,7,8-T_4CDD level in the 10 control subjects (i.e. non-Vietnam veterans), who thought that they had not been exposed to Agent Orange in any way, was 5.7 ± 3.1 ppt. This level, considered together with the Kingston and Ottawa average values of 12.4 ± 5.8 ppt and 8.6 ± 4.4 ppt respectively, suggests that 7-9 ppt may be considered a very crude estimate of the background level of 2,3,7,8-T_4CDD in the adipose tissue of the population.

Recent reports (Ryan et al., 1984 a & b, Graham et al., 1984, Schecter & Ryan, 1984) of a wide variety of PCDD and PCDF residues in adipose tissue of Canadian and U.S. populations indicate widespread exposure to these chemicals.

In the Canadian study (Ryan et al., 1984a) 66 human adipose tissue samples were obtained from accident victims from 5 regions across Canada, or patients deceased in Ontario hospitals. Samples were collected between 1972 and 1980. Analysis of these samples used modern technology capable of resolving specific isomers in the tetra- to octa- chlorinated isomer series. The results show that total PCDD plus PCDF residue levels range from 700 to 1700 ppt, mainly higher chlorinated PCDDs.

Characteristic profiles of about twelve 2,3,7,8-substituted PCDDs and PCDFs were found. These congeners include 2,3,7,8-T_4CDD; 2,3,4,7,8-P_5CDF; 1,2,3,7,8-P_5CDD; 1,2,3,4,7,8-H_6CDF; 1,2,3,6,7,8-H_6CDF; 1,2,3,6,7,8-H_6CDD; 1,2,3,7,8,9-H_6CDD; 1,2,3,4,6,7,8-H_7CDF; 1,2,3,4,7,8,9-H_7CDF;

1,2,3,4,6,7,8-H_7CDD and O_8CDD. Surprisingly, levels of 2,3,7,8-T_4CDF; 1,2,3,7,8-P_5CDF; 1,2,3,7,8,9-H_6CDF or O_8CDF were not detectable or at the limit of detection.

PCDFs represent about 10% of the total PCDD plus PCDD residues found. Levels of 2,3,7,8-T_4CDD and 2,3,4,7,8-P_5CDF only represent about 1-2% of the total residues. Three PCDD congeners, 1,2,3,6,7,8-H_6CDD; 1,2,3,4,6,7,8-H_6CDD and O_8CDD represent over 85% of the total residues found.

The relatively non-toxic O_8CDD congener formed 75-80% of the total residues found.

Inspection of the concentration distributions of PCDDs and PCDFs in human adipose tissue reported by Ryan et al., (1984a, b), Graham et al., (1984) and Schecter and Ryan (1984) indicate mean total PCDDs and PCDF residues of about 1,200 ppt. Application of the relative toxicity factors in Table 3.6.7C to these reported values suggests that this mean total PCDD plus PCDF (1,200 ppt) represents about 29 ppt of 2,3,7,8-T_4CDD toxic equivalents.

It is worth emphasizing the preliminary nature of the above results due to the small sample size and the lack of exposure data. Furthermore, comparison of PCDD levels in living persons and cadavers is an additional confounding factor. A joint Veterans Administration and Environmental Protection Agency Human Adipose Tissue survey, which will examine 15,000 samples in 1983-84 from broad geographical and age distribution groups, should put the above data in a better perspective.

5.1 INTRODUCTION

Exposure assessment is a complex, multidisciplinary analysis whose purpose is to describe the contact between the contaminant substance with one or more individuals in the population. A number of authors have proposed various approaches to exposure assessments and the data requirements for them (Nisbet, 1981; Reggiani, 1983; Kimbrough, 1983; Thibodeaux, 1983; Matsumura, 1982; Schmidt-Bleek, 1983; MacKay, 1982; Vaughan, 1983; Bennett, 1983; and Stevens, 1981).

All biological uptake information currently available is based on studies of laboratory animals orally or dermally exposed to high PCDD and PCDF concentrations. Consequently, in order to estimate the potential dose that could be absorbed by man (especially following inhalation), exposure models based on assumptions about rates of intake and uptake of estimated concentrations in air, water, soil or food must be used. Our environmental exposure (air, water, soil) and dietary intake (food and water) interact to form multiple exposure routes. Data on exposures via all these routes must be integrated to get the total exposure.

Thus exposure estimates from these routes (Section 5.2), and the exposure scenarios developed from them (Section 5.3) are of necessity hypothetical and the resulting hypothetical doses may not reflect actual quantities of PCDDs and PCDFs absorbed by man.

These exposure models were developed to assess the potential consequences of exposure to various levels of PCDDs and PCDFs in the Ontario environment. Worst case examples are used.

The hypothetical doses estimated are mixtures of PCDD and PCDF and not always as toxic as if they consisted of only 2,3,7,8-T_4CDD. In order to make more realistic estimates of toxicity, the exposures are calculated on the basis of the toxic equivalents approach discussed in Section 3.6.7.

The purpose of this exposure assessment is:

(i) to estimate current health risks in comparison with data on toxic effects;

(ii) to indicate the relative proportion of the daily dose from various sources to aid in setting standards for separate environmental media;

(iii) to give examples of how the toxic equivalents approach would be used in actual practice;

(iv) to compare routes of exposures with measured body burdens to determine whether all sources or routes of exposure have been taken into account;

(v) to identify potential problem areas for future investigation.

CAVEAT - the exposure assessments presented are hypothetical and should not be used to infer or come to conclusions about health risks to specific individuals or in specific geographical areas.

Generally the exposure assessment should utilize data on ambient concentrations of the contaminants from all routes of exposure (ingestion, inhalation or dermal), data on spatial and temporal variations in the exposure levels, data on the degree of absorption after exposure via each route and the distribution and pharmacokinetics of the contaminant within the body after absorption.

Data on the presence and concentration of the contaminant in body tissues are useful to compare with the estimated doses from the exposure assessment.

Frequently, as is the case with PCDDs and PCDFs, data on all the foregoing parameters are not available, so assumptions have been made in this exposure assessment.

Assumptions:

1. Ambient concentrations

   (i) Air - no ambient data available; therefore, an estimate was made using emission data from the SWARU municipal incinerator using models prescribed in Regulations under Environmental Protection Act, Ontario R.R.O., 1980.

   Reason - The only extensive data available are for the 3 Ontario incinerators analyzed. These are major facilities in Ontario. Only SWARU has had emissions characterized by congener type (Table 4.2.1.7B). Of the 3 incinerators studied these data represent worst case PCDD and PCDF emissions to the atmosphere.

Air Sources unaccounted for - in descending order of probable contribution to ambient concentration

- number of incinerators in one area
- wood combustion products
- apartment incineration
- chemical incineration
- power generation from fossil fuels
- biological waste incineration

(ii) Water - limited data on ambient concentration in surface water and drinking water available for Lake Ontario - since not detectable in drinking water only raw surface water used - this conservative approach overestimates intake

Water Sources unaccounted for - contaminated wells - in vicinity of chemical disposal sites or landfills where ash from incinerators is deposited - no data to indicate that this is a problem in Ontario

(iii) Soil - soil data from the study at SWARU are used to estimate exposure

Soil Sources unaccounted for - sites contaminated with incinerator ash or chemical wastes

(iv) Food - Lake Ontario fish represent a major source of 2,3,7,8-T_4CDD. Preliminary evidence suggests that poultry, pork and eggs may be important sources of higher chlorinated PCDDs, e.g. O_8CDD.

Food Sources unaccounted for - beef, milk, vegetables.

5.2 EXPOSURE PATHWAYS

The data to be used in this exposure assessment are:

Measured or estimated ambient levels in various environmental media (summarized in Sections 5.2.1 to 5.2.5); and

Biological monitoring data, i.e. levels of PCDDs and PCDFs measured in human adipose tissue (from Section 4.3.4.5.).

The measured and estimated distributions discussed below will be used in Sections 5.3.1 to 5.3.5 to develop exposure scenarios.

5.2.1 AIR

Data on atmospheric concentrations near chemical manufacturing/use facilities and waste sites in Ontario and in general are too limited for use in this exposure assessment. However, given the few sites in Ontario, they are not likely to be major contributors to ambient air levels of PCDDs/PCDFs in Ontario and their emission should not be significant.

Combustion sources have been reviewed in section 4.2.1.1. Reliable data are available from municipal incinerators and consequently only these data can be used in this exposure assessment.

Estimations of air quality in the vicinity of wood burning stoves have been made (Health and Welfare Canada and Environment Canada, 1983; Clement *et al.*, 1984). However, since only analysis of stack soot has been done rather than actual measurements of emissions from wood burning stoves, these estimates must be considered very preliminary, and are therefore not used in these exposure estimates.

Ambient air sampling for PCDDs and PCDFs has not been carried out in Ontario. Such sampling, as discussed in Section 4.3.1.3 has been done in only five places in the world.

Limited stack sampling data from municipal garbage and sewage sludge incinerators in Ontario are available.

The data from the SWARU tests of May, 1983 are used in this exposure assessment since PCDD/PCDF congener distributions are available. The data from Table 4.2.1.7C on the congener distribution of PCDDs and PCDFs from the SWARU incinerator and the data from Column 3, SWARU (May, 1983) Table 4.2.1.7B are used in determining the daily inhalation estimates in terms of 2,3,7,8- T_4CDD toxic equivalents. It is assumed in Table 5.2.1A that the congener distributions in the calculated ambient concentrations are the same as measured in the stack emissions (Table 4.2.1.7C).

The calculated 2,3,7,8-T_4CDD equivalent concentration in ambient air is shown in Table 5.2.1A using the relative toxicity factors discussed in Section 3.6.7.

Therefore the estimated maximum annual average ambient air concentration of 2,3,7,8-T_4CDD or its toxic equivalent to be used in this exposure assessment is 8.4 pg/m^3.

In Section 5.3.1, the average daily dose from this pathway is estimated.

TABLE 5.2.1A
CONVERSION OF TOTAL PCDD AND PCDF RESIDUES IN AIR NEAR THE ZONE OF MAXIMUM EXPOSURE NEAR SWARU TO 2,3,7,8-T_4CDD EQUIVALENTS

Maximum Annual Average Ground Level Concentration Distribution Col. 3 of Table 4.2.1.7B	Isomer Group Distribution[b] (Assuming distribution same as in stack emissions)	Toxicity Factors Relative to 2,3,7,8-T_4CDD (Table 3.6.7A)	2,3,7,8-T_4CDD Equivalents (pg/m^3)
pg/m^3 [a]	pg/m^3		2,3,7,8-T_4CDD[c] = 0.15
	T_4CDD 2.5	0.01	T_4CDD = 0.03
Total PCDDs 9.0	P_5CDD 2.4	0.1	P_5CDD = 0.24
	H_6CDD 2.3	0.1	H_6CDD = 0.23
	H_7CDD 1.0	0.01	H_7CDD = 0.01
	O_8CDD 0.75	0.0001	O_8CDD = 0.00007
Total PCDFs 19.0	T_4CDF 7.9	0.5	T_4CDF = 3.9
	P_5CDF 6.95	0.5	P_5CDF = 3.48
	H_6CDF 3.3	0.1	H_6CDF = 0.33
Total PCDDs and PCDFs 28.0	H_7CDF 0.65	0.01	H_7CDF = 0.007
	O_8CDF 0.2	0.0001	O_8CDF = 0.00002
		TOTAL 2,3,7,8-T_4CDD EQUIVALENTS =	8.4 pg/m^3

[a] May 1983 data doubled to account for operation of both boilers.
[b] Isomer group distribution (Table 4.2.1.7C).
[c] 2,3,7,8-T_4CDD is 6% of total T_4CDD (Ozvacic et al., 1984b).

5.2.2 WATER

From the point of view of human exposure assessment, drinking water and surface water need to be considered. No data on ground water concentrations of PCDDs and PCDFs are available for this assessment. However, the concentrations of PCDDs and PCDFs in ground water, that might impinge on drinking water supplies, should be considered in the risk management of PCDDs and PCDFs. Therefore, monitoring of ground water which may be contaminated should be considered in site specific areas.

MOE's earlier drinking water monitoring programs were targeted at total T_4CDD, and none was detected in drinking water in Ontario at detection limits down to 0.005ng/L. In raw water from Western Lake Ontario, total T_4CDD was detected in quantities ranging from 0.01 to 0.028 ng/L (Section 4.2.4.4) in only three out of 52 samples.

More recently M.O.E. surveys of surface water and drinking water included the tetra- to octachlorinated series of PCDD and PCDF congeners. In these surveys pg/L amounts of O_8CDD were detected in the raw water supply of 2 out of 15 waterworks. No PCDD or PCDF was detected in any of the treated drinking samples at the 0.005 ng/L detection level (Tosine et al., 1984).

Surface water concentration distributions from a very localized area of the Niagara River downstream of the S-area dump site indicate 0.001 - 0.20 ng PCDD/L and 0.005 - 0.05 ng PCDF/L (Table 4.2.4.1A). These levels of PCDFs and PCDDs cannot be considered typical of surface waters in Ontario.

In order to estimate the worst case exposure from water, surface water data from MOE drinking water monitoring programs are used. However, as no PCDDs or PCDFs have been detected in drinking water tested, this exposure pathway cannot be considered very important in its contribution to the total daily dose.

While 2,3,7,8-T_4CDD has not been positively identified in surface water, levels of this isomer in fish from Ontario waters indicate that it may be present at levels below present detection limits. On the assumption that 2,3,7,8-T_4CDD is N.D. (detection limit = 10 ng/kg) in most fish from Ontario waters (Section 4.3.4.4) and the B.C.F. ranges from 5,000 to 10,000 (Section 4.3.4.3), this isomer may be present at or below the 0.001 - 0.002 ng/L range. Consequently the worst case scenario for drinking surface water at 1.5 L/day suggests that daily intake of 2,3,7,8-T_4CDD from this source is in the 1.5 - 3.0 pg range.

Table 5.2.2A summarizes the calculated 2,3,7,8-T_4CDD toxic equivalent concentration from raw water analyses described in Section 4.2.4.4.

TABLE 5.2.2A
CONVERSION OF PCDD AND PCDF RESIDUES IN RAW WATER TO 2,3,7,8-T_4CDD EQUIVALENTS

Concentration Range (ng/L)	Toxicity Factors Relative to 2,3,7,8-T_4CDD (Table 3.6.7.A)	2,3,7,8-T_4CDD Equivalents (ng/L)
ND (0.001 - 0.002) (2,3,7,8-T_4CDD)*	1.0	0.001 - 0.002
ND (0.01) - 0.028 (T_4CDD)	0.01	0.0001 - 0.0003
ND (0.005) - 0.046 (O_8CDD)	0.0001	0.000005 - 0.00005
	Maximum Total 2,3,7,8-T_4CDD Equivalents	0.0024

* based on the assumption that levels in Ontario fish are related to non-detectable level in water as described in text.

In Section 5.3.2, daily dose from this pathway is estimated. The 2,3,7,8-T_4CDD toxic equivalent concentration for water for use in this exposure assessment is 0.002 ng/L.

5.2.3 SOIL

From the point of view of human exposure assessment, PCDD or PCDF contaminated soil near combustion sources, near waste sites, near chlorinated phenol and phenoxy herbicide production/use facilities or in areas where industrial waste has been applied to soil need to be considered.

Of the potential sources of soil contamination only soil near incinerators is considered in this assessment. Analysis of PCDDs and PCDFs in soil in the vicinity of Ontario incinerators is limited to data from SWARU.

For this exposure assessment, data from the estimated maximum ground level zone has been chosen (see Section 4.3.2.1). It is important to note that no relationship between PCDD and PCDF concentration or isomer group distribution with distance was found. Trace quantities of PCDDs and PCDFs are present in urban soil in general and in undisturbed soil, remote from urbanization. O_8CDD is the predominant congener found.

Table 5.2.3A summarizes the calculated 2,3,7,8-T_4CDD toxic equivalent concentration in soil with the highest measured PCDD and PCDF levels in the vicinity of an incinerator.

TABLE 5.2.3A

CONVERSION OF TOTAL PCDD AND PCDF RESIDUES IN URBAN ONTARIO SOIL TO 2,3,7,8-T_4CDD EQUIVALENTS

(Site 1260 m SW of SWARU, from Table 4.3.2.1B)

Isomer Group	Soil PCDD or PCDF concentration (pg/g)	Relative Toxicity Factor (Table 3.6.7A)	2,3,7,8-T_4CDD Equivalents (pg/g)
T_4CDD	n.d.	0.01	—
P_5CDD	580	0.1	58
H_6CDD	170	0.1	17
H_7CDD	390	0.01	3.9
O_8CDD	3,500	0.0001	0.35
T_4CDF	n.d.	0.5	—
P_5CDF	n.d.	0.5	—
H_6CDF	n.d.	0.1	—
H_7CDF	180	0.01	1.8
O_8CDF	n.d.	0.0001	—
TOTAL PCDD and PCDF =	4,820 pg/g	TOTAL 2,3,7,8-T_4CDD equivalents =	81.1 pg/g

The 2,3,7,8-T_4CDD toxic equivalent ambient soil concentration for use in this exposure assessment is 81.1 pg/g.

5.2.4 FOOD

Fish

Because of the bioaccumulation potential of T_4- and P_5CDDs and T_4- and P_5CDFs by fish, fish consumption is a very important factor from the point of view of human exposure assessment. It is important to note that the levels of 2,3,7,8-T_4CDD measured are from fresh water fish in very limited locales in Ontario, primarily western Lake Ontario.

Discussion of levels of PCDDs and PCDFs measured in Ontario fish are found in Section 4.3.4.4.

Other Food

Significant pg/g quantities of H_6-, H_7- and O_8CDDs have been detected in pork, poultry and eggs. (Section 4.3.4.3)

2,3,7,8-T_4CDD has not been detected in these other food items.

Table 5.2.4A summarizes the calculated 2,3,7,8-T_4CDD toxic equivalent concentration in selected dietary items based on data from Sections 4.3.4.3 and 4.3.4.4.

TABLE 5.2.4A

CONVERSION OF PCDD AND PCDF RESIDUES IN SELECTED FOOD ITEMS TO 2,3,7,8-T_4CDD TOXIC EQUIVALENTS

FOOD	PCDD/PCDF CONCENTRATION DISTRIBUTION (pg/g)	2,3,7,8-T_4CDD EQUIVALENTS (pg/g)
Fish	ND - 60 (20)* (100% 2,3,7,8-T_4CDD)	ND - 60 (20)
Chicken Fat	ND - 400 (60% O_8CDD)	ND - 8.8
Pork Fat	ND - 500 (85% O_8CDD)	ND - 0.8
Eggs	ND - 100 (100% O_8CDD)	ND - 0.01

*(20) - the MOE Fish Consumption Guideline for maximum levels of 2,3,7,8-T_4CDD in sportfish.

The 2,3,7,8-T_4CDD toxic equivalent dietary food concentrations for fish, chicken, pork and eggs, respectively, are 20 pg/g, 8.8 pg/g, 0.8 pg/g and 0.01 pg/g.

5.2.5 SUMMARY

From the discussions in Sections 5.2.1 through 5.2.4, multiple possible sources of exposure exist for people of Ontario.

Maximum estimated concentrations from the various sources are calculated as follows:

1) Ambient Air (Section 5.2.1)	- maximum annual average estimates of ground level concentrations range from 2 to 27 pg/m^3 of PCDDs and from 3 to 70 pg/m^3 of PCDFs in the vicinity of Ontario municipal waste and sludge incinerators - the calculated worst case 2,3,7,8-T_4CDD equivalent concentration for ambient air in this exposure assessment is 8.4 pg/m^3
2) Water (Section 5.2.2)	
Surface Water	- maximum ambient concentrations range from ND to 46 pg/L for PCDDs - the calculated worst case 2,3,7,8-T_4CDD equivalent ambient concentration for use in this exposure assessment is 0.002 ng/L
Drinking Water	- none (detection limit 0.005 - 0.01 ng/L).
3) Soil (Section 5.2.3)	- maximum measured soil concentrations near

	incinerators is 4640 pg/g of PCDDs and 180 pg/g of PCDFs - the calculated 2,3,7,8-T_4CDD equivalent soil concentration near incinerators is 81.1 pg/g of PCDDs plus PCDFs.
4) Food (Section 5.2.4)	
a) Fish	- concentrations in fish range from non-detectable to 60 pg/g of 2,3,7,8-T_4CDD. - only T_4- and P_5CDDs and T_4- and P_5CDFs are detected - note the maximum allowable concentration of 2,3,7,8-T_4CDD in sportfish for consumption in Ontario is 20 pg/g.
b) Other Food	- significant pg/g quantities of H_6-, H_7-, or O_8CDDs have been detected in pork, poultry and eggs - maximum 2,3,7,8-T_4CDD toxic equivalent concentrations are 0.01 pg/g (eggs), 0.8 pg/g (pork fat) and 8.8 pg/g (chicken fat).

5.3 ESTIMATED DOSES

As stated in Section 3.7, the recommended maximum daily intake (annual average) of total PCDD's and PCDF's should not exceed 10 pg 2,3,7,8-T_4CDD or its equivalent/kg body wt./day. This leads to the total allowed daily doses shown in Table 5.2.5A corresponding to various body weights.

TABLE 5.2.5A

RECOMMENDED MAXIMUM DAILY INTAKE OF 2,3,7,8-T_4CDD OR ITS EQUIVALENT OF PCDD's AND PCDF's

BODY WEIGHT (kg)	TOTAL DAILY DOSE OF PCDD's + PCDF's (pg)*
5	50
10	100
20	200
30	300
40	400
50	500
60	600
70	700
80	800
90	900

* based on 10 pg 2,3,7,8-T_4CDD or its equivalent of PCDDs + PCDFs/kg b.w./day

In order to be able to decide whether there is excess risk of developing adverse health effects, the total cumulative dose which an individual receives via various exposure routes needs to be estimated. This estimate can then be compared to the above allowed total daily doses corresponding to the body weight of the individual under consideration. This comparison of estimated total dose levels to recommended levels is the critical step in any exposure assessment.

In developing the exposure assessments for and assessing the health risks of PCDDs and PCDFs in Ontario, it is recommended in Section 3.6.7 that a method of accounting for the widely variable toxicity of different isomers of PCDDs and PCDFs be used. This recommendation is based on the results of monitoring data from Ontario discussed in Chapter 4, that shows that 2,3,7,8-T_4CDD (the most toxic isomer) comprises only a very small percentage of PCDDs and PCDFs to which the Ontario population is exposed with the exception of people eating some sport fish in the Great Lakes (especially in Lake Ontario). 2,3,7,8-T_4CDD is the most widely studied isomer. Therefore, it is recommended in Chapter 3 that 2,3,7,8-T_4CDD form the basis for recommended maximum daily intake and that estimates of the relative toxicity of the other isomers be made to facilitate conversion of their measured concentration to an equivalent concentration of 2,3,7,8-T_4CDD to compare with the recommended maximum allowable daily dose and therefore with the health risk.

Environmental concentration distributions in various media, bioavailability and pharmacokinetic processes were discussed previously. These data will now be synthesized into exposure tables, yielding estimates of daily doses via different routes of exposure. The tables can then be used for estimating total daily doses via all routes of exposure in any envisioned scenario or in a site-specific exposure assessment.

5.3.1 DAILY INHALATION

Inhalation of fly ash-bound PCDDs and PCDFs may be a source of exposure near incineration sources. Inhalation of soil-bound PCDDs, as suspended dust, is also possible in areas of soil contamination. However, no Ontario data on dust levels in air, whose sole source is contaminated soil, are available.

Annual average ambient concentrations and the worst case 2,3,7,8-T_4CDD equivalent concentration of PCDDs and PCDFs from municipal incinerators are discussed in Section 5.2.1.

Table 5.3.1A depicts possible daily doses based on the worst case estimate.

TABLE 5.3.1A

HYPOTHETICAL DAILY DOSE OF 2,3,7,8-T_4CDD OR ITS TOXIC EQUIVALENT FROM INHALATION

MEDIA/MATRIX	BREATHING RATE m^3/day	2,3,7,8-T_4CDD(b) EQUIVALENT CONCENTRATION pg/m^3	RETENTION(c) OF INHALED PARTICULATES %	ABSORPTION(d) THROUGH RESPIRATORY TRACT (%)	DAILY DOSE pg
AIR (estimated to contain gaseous and particulate bound PCDD's + PCDF's)	20	8.4	75	100	126

b - From Table 5.2.1A

c - Assumed that 75% of the inhaled particulates are retained in the body (ICRP, 1968).

d - Since no information is available 100% absorption/bioavailability was assumed.

5.3.2 DAILY INGESTION FROM WATER

Since PCDDs and PCDFs have not been detected in drinking water in Ontario (detection limit 10pg/L) the exposure table for water considers contaminated surface water. Table 5.3.2A was prepared using information from Section 5.2.2.

It is important to note that the absence of any PCDDs and PCDFs in the drinking water analyzed, indicates that ingestion from water is not likely to be a major route of exposure for the Ontario population.

TABLE 5.3.2A

HYPOTHETICAL DAILY DOSE OF 2,3,7,8-T_4CDD OR ITS TOXIC EQUIVALENT FROM WATER

MEDIA/MATRIX	DAILY CONSUMPTION (L/day)	2,3,7,8-T_4CDD(b) EQUIVALENT DISTRIBUTION ng/L (ppt)	ABSORPTION(c) FROM THE GI TRACT (%)	DAILY DOSE pg
Surface Water	1.5	0.002	90	2.7

b - Based on worst case assumption from Table 5.2.2A

c - Absorption from GI tract of 90% based on pharmacokinetic estimates in Table 3.2.7C

5.3.3 DAILY INGESTION OF SOIL (CHILDREN)

Small children are considered separately in this section. They may constitute a special group at risk when playing in or ingesting PCDD/PCDF contaminated soil. Table 5.3.3A considers daily ingestion from soil contaminated with PCDDs and PCDFs in the vicinity of municipal incinerators.

Absorption through the skin is estimated in Section 5.3.4.

TABLE 5.3.3A

HYPOTHETICAL DOSE OF 2,3,7,8-T_4CDD OR ITS TOXIC EQUIVALENT FROM DAILY INGESTION OF SOIL-BOUND PCCDs + PCDFs BY CHILDREN

MEDIA/MATRIX	AGE[a] GROUPS	DAILY[a] INGESTION PATTERN g	2,3,7,8-T_4CDD[b] EQUIVALENT CONCENTRATION pg/g	ABSORPTION[c] FROM THE GI TRACT (%)	DAILY DOSE pg
SOIL (contaminated with PCDDs + PCDFs from atmospheric deposition of fly ash near combustion sources or windblown from ash dumps)	1.5-5 yrs	1	81.1	5	4.06

[a] - Daily ingestion pattern by this age group (Kimbrough, 1984, personal communication).

[b] - Calculated from measured PCDD + PCDF soil concentrations in maximum ground level concentration zone from Table 5.2.3A.

[c] - Absorption estimates from Table 3.2.7C.

5.3.4 DAILY DERMAL

Absorption through the skin is possible following exposure from soil contaminated with chlorinated phenol and phenoxy herbicide related industrial wastes, as well as from soil contaminated with fly ash-bound PCDDs and PCDFs. Table 5.3.4A depicts daily doses possible from exposure to soil contaminated with PCDDs and PCDFs in the vicinity of a municipal incinerator. Assumptions and sources of information are listed in footnotes to the table.

TABLE 5.3.4A

HYPOTHETICAL DOSE OF 2,3,7,8-T_4CDD OR ITS TOXIC EQUIVALENT FROM DAILY DERMAL ABSORPTION

MEDIA/MATRIX	AGE[a] GROUPS	DAILY[a] DEPOSITION PATTERN OF SOIL ON SKIN g	2,3,7,8-T_4CDD[b] EQUIVALENT CONCENTRATION pg/g	ABSORP.[c] THROUGH THE SKIN (%)	DAILY DOSE pg
SOIL (contaminated with combustion-related PCDDs and PCDFs)	under 5 yrs	1 - 10	81.1	1	0.8 -8
	over 5 yrs	0.1 - 1	81.1	1	0.08-0.8

a - Daily deposition pattern of soil on skin by age group was assumed as shown (Kimbrough et al., 1984).

b - Calculated from PCDD & PCDF soil concentrations in maximum ground level concentration zone from Table 5.2.3A.

c - Absorption estimates from Table 3.2.7C.

5.3.5 DAILY INGESTION FROM FOOD

Exposure estimates from food shown in Table 5.3.5A below, are based on estimates from Section 5.2.4.

TABLE 5.3.5A

HYPOTHETICAL DOSE OF 2,3,7,8-T_4CDD OR ITS EQUIVALENT FROM DAILY INGESTION OF SELECTED FOOD ITEMS

FOOD	DAILY[a] CONSUMPTION (g/day)	2,3,7,8-T_4CDD EQUIVALENT[b] CONCENTRATION pg/g	DAILY[c] DOSE pg
Fish	14	20	280
Chicken Fat	5	8.8	44
Pork Fat	23	0.8	18
Eggs	40	0.01	0.4

a - Average daily Canadian consumptions were assumed (NRCC, 1981a)

b - from Table 5.2.4A

c - Absorption from the GI tract was assumed to be 100%.

5.3.6 Summary

Ranges of possible daily doses of 2,3,7,8-T_4CDD or its toxic equivalent of PCDDs and PCDFs have been estimated. It is unlikely that all of the maximum doses would be received by any one individual or group of individuals due to the variety of site-specific exposures which give rise to the estimated maxima.

In Section 5.4.1 the foregoing analysis is used to identify special groups who may be exposed to high concentrations of PCDDs and PCDFs.

5.4 EXPOSURE SCENARIOS, SPECIAL GROUPS AT RISK AND CURRENT LEVELS OF EXPOSURE

5.4.1 EXPOSURE SCENARIOS, SPECIAL GROUPS AT RISK AND CURRENT LEVELS OF EXPOSURE

It is clear, based on section 5.3 that, a large number of maximum total daily doses can be derived from the many combinations of exposure route - specific daily doses (using different sets of assumptions as to intake rates, absorption rates, contamination ranges, etc.).

Recently, two Canadian task forces identified and ranked the major pathways to human exposure, the special groups most likely to be exposed and the sites of most likely contamination. These conclusions are shown in Table 5.4.1A.

TABLE 5.4.1A

PATHWAYS, SPECIAL GROUPS AND SITES OF CONTAMINATION AS THEY RELATE TO HUMAN EXPOSURE*

REPORT OF THE JOINT HEALTH & WELFARE ENVIRONMENT CANADA EXPERT ADVISORY COMMITTEE ON DIOXINS (DEC., 1983)	DIOXINS IN CANADA: THE FEDERAL APPROACH INTERDEPARTMENTAL COMMITTEE ON TOXIC CHEMICALS (DEC., 1983)
-Direct human exposure through occupational and home use of chlorophenols, especially wood treatment and preservation -Air and air-borne particulates, in urban areas contaminated primarily from incinerators -Soil and sediments in urban areas contaminated primarily from incineration -Pharmaceutical and domestic products -Foods, almost entirely fish and unfiltered surface water near sources of dioxins -Other chlorophenol derived pest control products and point sources.	-Workers in the manufacture or formulation of chemicals containing dioxins -People using these chemicals directly e.g. in pest control spraying operations or at wood treatment facilities -Workers handling contaminated products e.g. sawmill workers, lumber suppliers or persons working in contaminated areas -High risk groups within the general population depending on their proximity to a source or their eating habits.

*- Order of pathways, special groups and sites of contamination as indicated and as ranked by the respective task forces.

Clearly it is impossible to envisage all the possible exposure scenarios affecting the general population of Ontario. Table 5.4.1B summarizes the exposure assessment for worst case estimates of doses of 2,3,7,8-T_4CDD or its toxic equivalent, developed in Section 5.3. These data in conjunction with the source assessment data summarized in Section 4.2.5. show that there are trends, which mirror the conclusions of the two task forces in Table 5.4.1A. These trends, based on a comparison of the estimated daily dose levels, received from each environmental compartment, (in Table 5.4.1B) to the total daily dose levels allowed by the standard (Table 5.2.5A), are (in order of significance):

- Some, but not all areas near combustion sources can be a source of exposure to total PCDDs and PCDFs via inhalation

- Ingestion of contaminated fish or other dietary items can result in exposure.

- Dermal exposure to and ingestion of urban soil by children

The above mentioned trends identify special groups at risk, in addition to occupational groups who are probably the most exposed.

The major CAVEAT must be repeated, i.e. the hypothetical or actual exposures experienced by Ontario citizens will mainly consist of PCDDs and PCDFs of low toxicity compared with 2,3,7,8-T_4CDD.

TABLE 5.4.1B

SUMMARY OF EXPOSURE ASSESSMENT (WORST CASE)

Environmental Compartment[1]	Total PCDD and PCDF Concentration Range [1]	2,3,7,8-T_4CDD Equivalents (pg)	Daily Dose (Absorbed) (pg) [2], [3]
Air (annual ambient ground level concentration)	28 pg/m^3	8.4 pg/m^3	126
Food - Fish	20 ng/kg	20 pg/g	280
- Poultry	400 ng/kg	8.8 pg/g	44
- Pork	500 ng/kg	0.8 pg/g	18
- Eggs	100 ng/kg	0.01 pg/g	0.4
Soil - Ingestion (children)	4820 pg/g	81.1 pg/g	4.0
- Dermal (children)			0.8 - 8
- Dermal (adult)			0.08 - 0.8
Surface water	0.002 ng/L	0.002 ng/L	2.7

1 - total PCDD + PCDF

2 - 60 kg individual processing 20 m^3 air, 1.5L water, 1g soil and Canadian average consumption factors (Table 5.3.5A)

3 - 2,3,7,8-T_4CDD toxic equivalents (from Table 3.6.7A)

A preliminary estimate of current levels of exposure of the general population is possible through consideration of the background levels of PCDDs and PCDFs in terms of 2,4,7,8-T_4CDD toxic equivalents in human adipose tissues. These levels were discussed in section 4.3.4.5 where it was recommended that 29 ppt is a reasonable estimate of the background level of 2,3,7,8-T_4CDD toxic equivalent in the adipose tissue of the human population. As depicted briefly in Table 5.4.1C and in more detail in Appendix I, the current level of exposure to 2,3,7,8-T_4CDD toxic equivalents by the population of Ontario is estimated to be 7.2 pg/kg.bw/day.

Summary

Compensating for the variable toxicity of the mixtures of PCDDs and PCDFs to which the population of Ontario may be exposed by using 2,3,7,8-T_4CDD toxic equivalents, various exposure scenarios have been estimated. From this analysis special groups at risk can be identified: people living near some, but not all combustion sources, who inhale PCDDs and PCDFs adsorbed on airborne particulates, people eating contaminated fish or other dietary items and children dermally exposed to or ingesting urban soils.

Only the contaminated fish contains a high proportion of 2,3,7,8-T_4CDD in the total PCDDs/PCDFs measured. In the other scenarios, comparison of the calculated dose from total PCDDs and PCDFs with the recommended maximum daily dose based on 2,3,7,8-T_4CDD results in an overestimation of the health risk.

Information from source assessment (Section 4.2.5) and two Federal reviews of pathways of human exposure indicate that occupational groups involved in the formulation or use of chemical products containing PCDDs or PCDFs are probably the most highly exposed.

The current level of exposure of the population of Ontario to 2,3,7,8-T_4CDD or its toxic equivalent from all sources, based on analysis of residues in human adipose tissue, is estimated to be 7.2 pg/kg body weight/day.

TABLE 5.4.1C

ESTIMATION OF CURRENT LEVELS OF EXPOSURE

1. Estimate "average" body burden ($\bar{X}_{ss}$) at steady state from measured adipose tissue levels and estimated liver and other body tissue levels.[a]

2. Assume: Time interval (t) between exposures: Daily (1 day)
   Half life ($t^1/_2$) of 2,3,4,7,8-T_4CDD in human body[b] : 365 days
   Amount absorbed : 100%.

3. Use equation:
   Current Level of Exposure = Total Daily Dose = $\frac{\bar{X}_{ss} \times t}{1.443 \times t^1/_2}$ (Equ.1)

   and values:

   $\bar{X}_{ss}$ = 264.3 ng 2,3,7,8-T_4CDD toxic equivalents/70 kg man.
   t = 1
   $t^1/_2$ = 365 days.

4. Current level of exposure = 7.2 pg 2,3,7,8-T_4CDD toxic equivalents/kg.bw/day

[a] - See Appendix I for details.
[b] - See Table 3.2.7B and Section 3.2.7.

5.4.2 SITE-SPECIFIC EXPOSURE ASSESSMENT

The development of exposure scenarios is useful in identifying special groups at risk and in estimating current levels of exposure (Section 5.2.6). This assessment, however, does not lead to making decisions in a particular situation such as: 1) is this a 'hot spot'; 2) is limitation of exposure required; or 3) is immediate action vs. long term action needed?

Because of the complexities and variability of circumstances already described in sections 5.2.5 and 5.2.6 decisions on risk management will necessarily be based on site-specific assessments and environmental hearings on major future projects.

It is recommended that part of such assessments and hearings should be site-specific exposure assessment. Recently published examples, summarized in Table 5.4.2A, provide useful guidelines for such assessments.

These assessments, in most cases, are done in conjunction with environmental monitoring of air, water, soil, etc., aiding the credibility of the assessment substantially. The presently available analytical methods of monitoring may be greatly reinforced in the future by biological methods of monitoring as detailed in Section 6.2.

TABLE 5.4.2A
GUIDELINES FOR SITE-SPECIFIC EXPOSURE ASSESSMENT

TYPE OF SITE	EXPOSURE PATHWAYS or FATE TO EXPOSURE PATHWAYS ADDRESSED	REFERENCE
Municipal Incinerator	1. Inhalation. 2. Uptake from contaminated food, grown in maximum fallout zone, i.e. milk.	(Olie *et al.*,1983)
Municipal Incinerator	1. Inhalation. 2. Uptake from contaminated food, grown in maximum fallout zone, i.e. milk, meat, vegetables, Scenario considers: Deposition rate onto grass Half-life on grass Daily roughage ration of cow Absorption of pollutant by cow Pollutant content of milk, meat, etc.	Swiss Federal Office for Environmental Protection, 1982)
Municipal Waste Resource Recovery Facility	1. Inhalation.	(EPA, 1981) (EPA, Draft III, 1983)
Municipal Waste Resource Recovery Facility	1. Inhalation.	(Consolidated Hearing Board, 1983)
Area of Soil Contaminated With 2,3,7,8-T_4CDD of 2,4,5-TCP origin	1. Inhalation. 2. Ingestion (children) - Considers age specific amount of soil ingested and absorption. 3. Dermal exposure - Considers age specific amount of soil deposited on skin and absorption. 4. Uptake from contaminated food grown in contaminated soil. 5. Concentration of 2,3,7,8-T_4CDD in beef, pork and milk fat, resulting from a certain concentration in soil using PBB's as surrogate. 6. Estimation of current soil concentration since actual measurement, using an equation and assuming a 12 year half-life in soil.	(Kimbrough *et al.*, 1984)
Waste Site Containing 2,4,5-T Related Waste Buried in Soil Next to Creek and Pond	GENERAL OFFSITE TRANSPORT OF 2,3,7,8-T_4CDD - water in creek, - vaporization from soil surface, - vaporization from landfill cells, - vaporization from pond surface, - windblown dust.	(Thibodeaux, 1983)
Solid Waste Reduction Unit (Swaru)	Inhalation - - Effect of various plant operating conditions on stack emissions.	(Ozvacic *et al.*, 1984a)
Solid Waste Reduction Unit (Swaru)	Ingestion, Inhalation - - Soil contamination in the fallout zone.	(McLaughlin and Pearson, 1984)

5.4.3 IMPLICATIONS FOR RISK MANAGEMENT

"Decision makers have a responsibility to articulate their total evaluation in such a way that the public can appreciate the fact that an indication of non-zero risk is not necessarily a significant risk" (Barnes, 1983).

The following conclusions regarding the estimated exposure assessment and the recommended maximum allowable daily intake derived in this document, may be useful to decision makers involved in setting the standard for the various environmental media and in risk management:

1. Exposure is not consistent over a lifetime. For example:

    (a) People and children would not be exposed to contaminated soil on a continuous daily basis (i.e. during snow cover and inclement weather).

    (b) Indoor particulate and soil levels may be 10 to 100- fold lower than outdoor levels. Considerable time is spent indoors.

    (c) Contaminated surface water, elevated levels of PCDDs in fish and some food items are very low probability situations.

    (d) People in the vicinity of combustion sources are not continually exposed to maximum ground level concentrations of emissions.

2. Estimates of the current levels of exposure do not exceed the recommended maximum allowable daily intake.

3. Bioanalytical methods should provide simple, cost-effective monitoring and early warning tools for present and potential future problems.

4. Development of the environmental standards requires consideration of:

    (a) the recommended maximum allowable daily intake of PCDDs + PCDFs equivalent to 10 pg 2,3,7,8-T_4CDD/Kg.bw./day;

    (b) establishment of protocols for methods of analytical measurement in the various matrices (i.e., air, soil, water, etc.), see Sections 5.3.1 and 5.3.2;

    (c) the development of standards for allowable levels in the various matrices (i.e., air, soil, water);

    (d) development of monitoring programs of emissions from stacks, waste sites, waste waters, etc.

5. Possible approaches for development of standards for the various matrices are:

    (a) Apportion the maximum allowable daily intake (ADI) of 10 pg 2,3,7,8-T_4CDD equivalent/kg.b.w./day equally among the various media: soil, water, food, air (Birmingham, 1983) (Appendix III);

(b) Indicate different maxima for the various media so that no one route of exposure can be allowed the total maximum ADI;

(c) Allow all the ADI to be apportioned to one route of exposure;

(d) Assume water accounts for 10% of the ADI;

(e) Use current limits of detection to apply more stringent levels of control;

(f) Not to measurably increase the PCDD and PCDF content of an environmental matrix above current background levels;

(g) Use guidelines proposed by other regulatory agencies.

6. Combinations of the approaches discussed in point 5 above might be used eg. option (c) could be applied to drinking water and option (f) to air, soil and diet (fish consumption).

7. Another consideration in apportioning the maximum allowable daily intake is to consider special groups at risk e.g. occupationally exposed persons as compared with other members of the population who have little or no exposure to PCDDs or PCDFs.

6.0 ANALYTICAL METHODS FOR MONITORING PCDDs AND PCDFs

6.1 PCDDs AND PCDFs: REVIEW OF ANALYTICAL METHODS

Introduction

Few refinements in the analysis of PCDDs and PCDFs have occurred since the review published by the National Research Council of Canada in 1981, (NRCC, 1981b). A 1981 review concerning the state-of-the-art of PCDD from combustion sources (ASME, 1981) and two 1983 reviews (Crummett, 1983; Tiernan, 1983) also describe the analysis of PCDD/PCDF compounds in various types of samples. Recent reports of the analysis of these substances at parts-per-quadrillion (ppq) concentrations in water have achieved these limits by high concentration factors, not by improvements in the basic analytical methodology (Tosine et al., 1984).

It is not the purpose of this review to be comprehensive in coverage of methods that are well documented elsewhere (NRCC, 1981; ASME, 1981; Crummett, 1983; Tiernan, 1983). An attempt has been made to give greater coverage to specific areas of analysis, such as sampling, extraction and clean-up, that are generally covered in less detail than the instrumental techniques which are used. Emphasis will be on those techniques demonstrated to be the most effective. Quality control procedures, quantification methods and limitations of existing methods will be examined. General review of various aspects of the analytical process will be followed by detailed discussion of the methodology developed for different types of samples.

Sampling Methods

Although the importance of proper sampling procedures is obvious, this aspect of analysis has often been overlooked. Few studies of sample homogeneity or sampling reproducibility have been reported. One reason for this is the cost of analysis, which may be $1,000.00 or more per sample. Nevertheless, estimates of the total environmental input of PCDD/PCDF compounds from a few analyses, often non-replicated, may be incorrect by large amounts. Such errors may be acceptable in general surveys designed to identify specific sites for more detailed testing; however, a higher degree of confidence is required if the data are used for setting exposure guidelines or for regulatory purposes.

Sample Extraction

Many different solvent systems have been employed. Although aromatic solvents such as benzene, toluene, and xylene are clearly superior on a solubility basis to other solvents that have been used, including methylene chloride and pentane or hexane, the most effective solvent for a specific application depends on the type of sample. Factors such as emulsion formation and penetrating ability of the solvent in the sample matrix are as important as solubility of PCDD/ PCDF compounds.

Common extraction techniques such as Soxhlet extraction and ultrasonic extraction are generally used for particulates. Liquid-liquid extraction or sorbent trapping followed by extraction from the sorbent material have been applied to aqueous samples. Vapors in stack emissions are collected by a combination of cold-trapping and sorbent cartridges, and extracted as for aqueous samples. Jar extraction methods, in which the sample and solvent are shaken in a jar for a short period of time, are sometimes suitable for survey-type work if PCDD/PCDF are in the parts-per-billion (ppb) range.

To achieve optimum recovery of PCDD/PCDF from the sample usually requires destruction or modification of the sample matrix. For biological samples, acid or base digestion is employed, followed by liquid-liquid extraction. Treatment using concentrated base may result in decomposition of some of the higher chlorinated congeners, therefore acid digestion should be employed, unless only lower chlorinated PCDD/PCDF congeners are determined (Albro and Corbett, 1977). Acid treatment of particulate samples, such as the electrostatic precipitator fly ash from municipal incinerators, has been shown to give higher recoveries compared to non- treated particles (Lustenhouwer et al., 1980). Aqueous and vapor-phase samples do not need special treatment other than clean-up.

Sample Clean-up

Much work has been reported concerning the development of techniques to separate PCDD and PCDF congeners from co- extractives that may interfere in the instrumental determination. The potential problem of interferences is especially important at ultra-trace levels (ppt or lower concentrations), even when a highly selective detector such as a mass spectrometer is employed. Most PCDD/PCDF analyses are for environmental samples which contain hundreds of other organic compounds, many of which are initially present at hundreds to thousands times higher concentrations than the PCDD/PCDF. Benefits of using clean-up procedures before GC- MS analysis are that they:

1. reduce the possibility of obtaining false positives by eliminating most known interferences - a specific clean- up procedure combined with GC-MS detection of 2 or 3 characteristic ions in the proper ratios at correct retention times can be considered positive evidence for the presence of PCDD/PCDF,

2. increase signal to noise ratio for GC-MS detection by removing substances that contribute to the general background response,

3. increase sample throughput by extended column life and less-frequent mass spectrometer ion source cleaning.

Most reported clean-up methods are based on column chromatography using a variety of packing materials. Commonly used adsorbents are silica gel, alumina, and celite 545. These materials are used to separate PCDD/PCDF congeners from mirex, DDE and similar pesticides and polychlorinated biphenyls (PCB). The retention of these substances on silica gel generally depends on the degree of chlorine substitution, with the higher chlorinated species being the first to elute (Dolphin and Wilmot, 1978). The range of retention of PCDD/PCDF congeners on silica gel overlaps that of PCB compounds and other potential interferences, such as polychlorinated naphthalenes. On alumina, however, the retention of chlorinated compounds increases with increasing chlorine substitution.

Therefore, the combination of silica gel-alumina is very effective for isolating PCDD/PCDF congeners from common interferences. Lipids from extracts of biological substances are removed by concentrated H_2SO_4 on silica gel, and sulphur compounds can be removed using $AgNO_3$. Celite 545 is sometimes used instead of silica gel. Florisil has also been used in combination with alumina and silica gel or by itself for extracts that are not expected to contain PCB, chlorinated naphthalenes, pesticides, or similar substances.

High performance liquid chromatography (HPLC) has been found to be very effective for clean-up of environmental extracts for PCDD/PCDF analysis. Most reported applications use HPLC after column chromatographic clean-up to effect the isomer-specific analysis of 2,3,7,8-T_4CDD. The combination of column chromatography/HPLC/high resolution gas chromatography for the unambiguous quantitative determination of 2,3,7,8-T_4CDD can be used in extracts of environmental samples, even when all 22 TCDD isomers are present (Lamparski and Nestrick, 1980). HPLC can also be used instead of liquid chromatography for the separation of PCDD/PCDF congeners from other substances, such as PAHs, in some types of samples (Karasek et al., 1983).

Instrumental Analysis

Gas chromatography combined with low or high resolution mass spectrometry is necessary for the quantitative determination of picogram amounts of PCDD/PCDF congeners in environmental extracts. Although packed or wall-coated open tubular (WCOT) columns may be employed, fused silica WCOT columns were used in most recent studies. This is due to their ease of installation, high performance and ruggedness. Cross-linked fused silica columns are durable enough to withstand back-flushing with aromatic solvents, and can withstand higher GC temperatures than glass WCOT or non-cross-linked fused silica WCOT columns. By using several WCOT columns, all 22 T_4CDD isomers can be separated (Buser, 1980). No single column, however, has been developed that can accomplish this separation.

The use of low resolution (LRMS) versus high resolution mass spectrometry (HRMS) has been discussed (Hummel and Shadoff, 1980). LRMS can be used to positively identify PCDD/PCDF congeners if the proper wet chemical sample workup has been performed. This will vary according to the type of sample, but generally includes some liquid chromatography and/or HPLC. An exception is if the concentrations of PCDD/PCDF are large compared to concentrations of other compounds that respond to the specific masses monitored and in the same GC elution region. In these situations, often found for extracts of fly ash from municipal incinerators, no clean-up may be required. Detection limits, however, will almost always be lower for cleaned-up extracts. Confirmation of PCDD/PCDF compounds should include the following:

1. Observed peak appears in correct elution time region.

2. Response is at least 3x average or root-mean-square background noise level.

3. Exact correspondence of peaks at 2 or 3 characteristic masses.

4. Correct area ratios of corresponding sets of characteristic masses (theoretical ±15%).

Isomer patterns can also aid identification, since samples taken from the same source usually have similar patterns (Ozvacic et al., 1984a). When determinations are made close to the instrumental detection limit, isomer ratios may be farther from the theoretical values than is usually observed, and in these situations isomer patterns may still indicate the presence of a specific group of PCDD/PCDF congeners. For specific isomer determinations, the identification criteria include exact retention time matching of native and spiked, isotopically labelled internal standard. These criteria are discussed in a recent review (Karasek and Onuska, 1982).

Of the other instrumental techniques used in PCDD/PCDF analysis, the ones which are most promising are negative chemical ionization MS (NCIMS) (Hass et al., 1978) and MS-MS (Voyksner et al., 1983). NCIMS has lower detection limits for higher degrees of chlorination, and can be made specific for 2,3,7,8-T_4CDD (Hass and Friesen, 1979). Because overall detection limits for all PCDD/PCDF compounds in NCIMS are not substantially lower than can be obtained by standard electron-impact (EI) MS, NCIMS is not likely to supplant regular EIMS as the routine detection method for the PCDD/PCDF. For ultra-trace GC-MS analysis, about 1 to 10 picograms of a pure standard injected into the GC should give a distinguishable peak (at least 3x background level) which has correct ion ratios for 2 or 3 ions monitored. These limits can be achieved with most conventional GC-low resolution MS instruments.

The technique of MS-MS has two important features for the analysis of PCDD/PCDF congeners. Since mass filtering is performed twice, the possibility of more definitive identification exists. Also, the more specific detection may reduce the degree of sample clean-up necessary. To date, however, no instrumental technique has demonstrated the ability to perform unambiguous detection and quantification of PCDD/PCDF in all types of complex environmental extracts at ppt levels without some form of clean-up. Considering the initial high cost of MS-MS instrumentation, there are presently no significant advantages to using this method over regular GC-LRMS. When more definitive analysis is required, HRMS can be used.

Quality Control Procedures

A large portion of the quality control work reported for PCDD/PCDF analysis has been dedicated to the prevention of false positives. Reasons for this are the very low picogram amounts which must be detected, and the potential impact of a positive finding from a new source, regardless of the actual concentration present. Elimination of false positives depends upon the removal of interfering compounds from the sample extract, prevention of external contamination of the sample, and following strict criteria for assigning GC- MS peaks as PCDD/PCDF congeners.

Criteria for GC-MS identification and procedures for sample clean-up have already been discussed. Prevention of external contamination is assured by analyzing proper blanks and by thorough glassware cleaning. The extent to which these procedures are required depends upon the type of sample analyzed and the detection limit which must be attained. Procedures used at the Ontario Ministry of Environment for ultra- trace PCDD/PCDF analysis in water samples include the following (Tosine, Clement and Hunsinger, 1984):

1. Sample Container Blank: rinse sample containers with extracting solvent, save final rinse for GC-ECD or GC-MS analysis - repeat cleaning until no peaks (GC-ECD) or no PCDD/PCDF response (GC-MS) observed.

2. Glassware Blank: at end of glassware cleaning procedure, collect final solvent rinse to analyze for PCDD/ PCDF as in #1 above.

3. Procedure Blank: take PCDD/PCDF contamination-free water to sampling site, pour water into pre-cleaned empty containers during sample collection and at same physical location as samples are collected; treat this blank in the same manner as samples throughout the analysis procedure.

4. Dedicated Equipment: all glassware is dedicated according to the matrix: fish, water or incinerator samples and is cleaned, stored and used separately; re-claimed reacti-vials used for sample storage are cleaned and checked before re-use as previously explained for other glassware; vials formerly containing high levels of PCDD/PCDF such as those used to store incinerator samples and standards are treated apart from other vials; syringes used to inject water sample extracts are not used for any other samples or standards.

5. **Alumina Activity Check:** activity of alumina column is checked before sample clean-up for each new batch prepared by eluting a standard mixture containing a representative compound for each PCDD congener as well as TCDF, OCDF; eluant is collected, concentrated and analyzed by GC-ECD to ensure the proper elution pattern is obtained; new batches of alumina columns are prepared if results are not satisfactory.

6. **Isolated Laboratory:** water samples are physically prepared in a laboratory apart from where other samples (especially incinerator) are treated; a positive air pressure system reduces the possibility of contamination from outside dust.

7. **Instrumental Detection Limit:** the needed instrumental detection limit is demonstrated by analyzing low picogram-level standards, not by extrapolation down from a concentrated standard.

8. **HRMS Confirmation:** selected samples having PCDD/PCDF detected by GC-low resolution mass spectrometry are analyzed by GC-high resolution mass spectrometry at a resolution of at least 10,000 to confirm the presence of PCDD/PCDF.

One of the most important quality control procedures is the addition of isotopically labelled compounds to the sample before extraction. These internal standards can be used to correct for sample losses in the extraction and clean-up stages of analysis. 13C-labelled 2,3,7,8-T_4CDD and 37Cl-2,3,7,8-T_4CDD are the most commonly used isomers. If analysis of the entire range of congeners from T_4- to O_8CDD is performed, 13C-O_8CDD or 37Cl-O_8CDD should also be added, since the higher chlorinated congeners do not necessarily behave in exactly the same manner as the lower chlorinated species. For the isomer-specific analysis of 2,3,7,8-T_4CDD, an important criterion is the exact correspondence of retention times for the added ^{13}3C-2,3,7,8-T_4CDD or ^{37}Cl-2,3,7,8-T_4CDD and the native 2,3,7,8-T_4CDD.

Quality control checks by analysis of split or replicate samples by 2 or more laboratories have been performed in only a few reported studies. Interlaboratory investigations have been reported for fish (Ryan et al., 1983b) and water (McMillin et al., 1983) analysis. In the fish sample round-robin analysis, a relative percent standard deviation (%RSD) of 19% was obtained for the analysis of 4 different samples by 8 laboratories. Average %RSD for recovery of labelled T_4CDD was 25%. In most cases, interlaboratory results were comparable, even though different instruments and procedures were used. Results of an initial review of a validation study for PCDD in water gave the average %RSD for analysis of 2,3,7,8-T_4CDD in water to be 16% for concentrations ranging from 20 to 200 ppt (McMillin et al., 1983). Ten laboratories participated in this investigation. Results between different laboratories were generally comparable. Similar studies need to be performed on particulate samples such as precipitator fly ash. For ppt analysis of PCDD/PCDF compounds, a relative %RSD of about 20% for interlaboratory testing seems to be the state-of-the-art with current techniques.

Quantitative Analysis

The precision and accuracy of PCDD/PCDF quantitative data are now receiving more attention, since fundamental analytical methodologies have been developed for most types of environmental samples.

A fundamental problem has been the availability of pure PCDD/PCDF standards. Until recently, only a few of the 210 possible PCDD/PCDF compounds were commercially available. Also, only a few isotopically labelled standards needed for recovery studies and as internal standards could be obtained. Quality control of these standards is questionable. This situation is rapidly changing, and representative standards of each PCDD/PCDF congener group are now available.

Studies measuring concentrations of only 2,3,7,8-T_4CDD generally produce more accurate data than investigations which measure concentrations of the range of congeners from the tetra- to the octachlorinated PCDD/PCDF, since pure 2,3,7,8-T_4CDD standards are available. For PCDD, at least one representative compound of each congener may be obtained commercially. Only a few of the PCDF congeners, however, are available.

Response factors for all members of a congener group are generally assumed to be the same for quantification by only one representative compound. PCDF congeners are usually quantified using the corresponding PCDD standard. In some studies where only a few standards were available, response factors for missing congeners were interpolated from available standards. Table 6.1A gives relative GC-MS response factors for selected PCDF standards (Rappe et al., 1983b). These data show that interpolation of response factors for quantification should not be performed using 1 or 2 standards, and that even response factors within a specific congener group may vary considerably. A more recent study (Kuroki et al., 1984) also confirms these findings. It has also been observed that PCDF compounds generally have larger response factors for electron impact MS than corresponding PCDD compounds (Coburn, 1983; Clement and Tosine, 1983).

TABLE 6.1A

RELATIVE GC-MS RESPONSE FACTORS* FOR PCDF COMPOUNDS: NORMALIZED TO 2,3,7,8-T_4CDF = 100

COMPOUND	RELATIVE RESPONSE ELECTRON IMPACT
2,3,7,8-T_4CDF	100
1,3,6,8-T_4CDF	130
1,2,6,7-T_4CDF	80
1,2,4,6,8-P_5CDF	120
2,3,4,6,8-P_5CDF	50
2,3,4,6,7,8-H_6CDF	60
1,2,3,4,7,8-H_6CDF	40
1,2,3,4,6,7,8-H_7CDF	35
O_8CDF	70

* - from (Rappe et al., 1983b)

General Methods for Various Sample Types

A. Chlorophenol Formulations and Related Products (Firestone et al., 1972; Buser and Bossardt, 1976)

1. Phenolic compounds removed by alkaline extraction.
2. PCDD/PCDF extracted by petroleum ether.
3. Final clean-up on alumina column.
4. Some methods include concentrated H_2SO_4 rinsing of the PCDD/PCDF fraction from the alumina column.

B. Incinerator Particulate Samples (Kooke et al., 1981; Taylor et al., 1983, Ozvacic et al., 1983)

1. Acid-treat particulates with dilute HCl.
2. Filter particulates, air-dry.
3. Extract PCDD/PCDF by Soxhlet using toluene for about 16 hr.
4. Concentrate extract, remove oxidizable compounds on a column containing silica, H_2SO_4 on silica, and NaOH on silica.
5. Remove other interferences on a second dual-column system consisting of $AgNO_3$ on silica (first column), and basic alumina (second column).

C. Fish and Tissue Samples (Tosine et al., 1983a; Hummel, 1977)

1. Dissolve tissue in concentrated HCl.
2. Extract with hexane.
3. Initial clean-up; combination of silica, H_2SO_4 on silica, NaOH on silica packed in a single column.
4. Second column clean-up; $AgNO_3$ on silica followed by basic alumina.
5. PCDD fractions further cleaned-up by normal phase silica HPLC followed by reverse-phase HPLC.

D. Soils and Sediments (Balasso et al., 1983)

1. Air-dry sample.
2. Soxhlet extract for about 16 hr. using toluene.

3. Concentrate extract, clean-up on column consisting of silica, H_2SO_4 on silica, and NaOH on silica.
4. Further clean-up using a dual-column with $AgNO_3$ on silica (top column) and alumina.

E. Industrial and Municipal Water Supplies (Tosine et al., 1984; Wong et al., 1983)

1. Liquid-liquid extract sample with methylene chloride, pentane or toluene.
2. Clean-up columns as described in D.

Limitations to Existing Methodology

A great deal of additional research is needed to determine the validity of current methodology. Standardized methods for reporting recoveries and determining detection limits must be developed if data from different sources are to be properly evaluated and compared. Many more studies of sampling methods, to study reproducibility, precision and accuracy are needed.

The principal limitations to achieving better quantitative data at sub-parts-per-billion levels are the availability of reliable standards and the wet chemical workup. Instrumental capabilities have now reached the point where parts-per-trillion and in some cases parts-per-quadrillion detection limits can be reached, depending upon the sample concentration factor. High inter-laboratory variations in quantitative data can often be attributed to differences in sample extraction/cleanup and standardization.

6.2 BIOLOGICAL ANALYSIS

Several methods of analysing various media for PCDDs and PCDFs using biological techniques are currently under development. These areas of biological analysis are: radioimmunoassay, AHH induction, cytosol receptor assay, tissue cultures and whole animal tests. Each of these tests is based on the biological/biochemical properties of PCDDs and PCDFs. Current limitations include limited experience and validation and lack of specificity and sensitivity (when compared with current analytical techniques). Potential advantages of these tests include low cost, large-scale screening ability and biological realism.

A review of early papers on radioimmunoassay, AHH induction assay and cytosol receptor assay (NRCC, 1981b) suggests that the radioimmunoassay has sensitivity and specificity problems related to the purity of antigen and antisera available.

AHH induction assays are based on the ability of PCDDs and PCDFs to induce high levels of activity of hepatic microsomal cytochrome P-450 monooxygenases (Knutson and Poland, 1982). Traditionally, in vivo assays have involved administration of PCDDs and PCDFs to rats followed by several days to allow for induction of the enzymes. The rats are then sacrificed, liver microsomes extracted and the enzyme activity estimated using benzo(a)pyrene (AHH) as substrate. The activity of other enzymes that function in sequence with cytochrome P-450 monooxygenases eg. epoxide hydrolase, 7-ethoxycoumarin 0-deethylase or 7-ethoxyresorufin-O-deethylase can also be used (Sawyer and Safe, 1982; Sawyer et al., 1983; Bandiera et al., 1984).

A more promising _in vitro_ variant of this assay measures induced levels of these enzymes in cultures of rat hepatoma cells. (Bradlaw and Casterline, 1979). Primary cultures of rat hepatocytes have also been used (Knutson and Poland, 1980a; Steward and Byard, 1981).

AHH induction has been used to estimate the "2,3,7,8-T_4CDD equivalent" activity of PCB and PCDFs mixtures (Sawyer and Safe, 1982, Bandiera _et al_., 1984), food extracts (Bradlaw and Casterline, 1979), extracts from incinerator flyash (Sawyer _et al_., 1983) and soil from Times Beach, Missouri (McConnell _et al_., 1984). Generally this assay measured biological activity at levels many-fold higher than the chemically analyzed 2,3,7,8-T_4CDD content reflecting the presence of other biologically active PCDDs and PCDFs and possibly other polychlorinated aromatic hydrocarbons which induce AHH activity.

The AHH induction assay is not as specific or as sensitive as current chemical analyses and therefore extraction and concentration of PCDDs and PCDFs in environmental media is required. However, this assay does measure a biological response to PCDDs and PCDFs and can be calibrated with 2,3,7,8-T_4CDD.

The cytosol-receptor assay relies on displacement of radioactively labelled 2,3,7,8-T_4CDD from a specific cytosol protein by unlabelled PCDDs or PCDFs (Poland _et al_.,1976). The effective concentration causing 50% displacement of the radioactively labelled 2,3,7,8-T_4CDD (EC_{50}) is calculated from calibration curves. This assay has been applied to mixtures of PCBs and PCDFs

(Bandiera *et al*., 1984) and extracts from municipal incinerators (Sawyer *et al*., 1983). This method appears to be less sensitive than AHH induction due to non-specific binding problems.

The *in vitro* hyperkeratinization assay of Knutson and Poland (1980b) is based on the induction of keratin protein synthesis by PCDDs and PCDFs in a co-culture of irradiated 3T3 mouse cells and XB cells cloned from a mouse teratoma. The induced keratin protein is visualized and quantified by staining the cultures several weeks after exposure to the PCDD or PCDF extract. This test has been applied to soot from the PCB fire in the Binghampton State Office Building (Gierthy and Crane, 1983). The cells responded in a semiquantitative fashion to soots from various parts of the building. A major problem with cell cultures of this kind is maintenance of stable cell lines.

The bioassays that show the most promise as analytical tools to measure the "2,3,7,8-T_4CDD equivalent" activity of environmental extracts are the AHH induction and cytosol-receptor assays. Both assays are receptor-mediated and therefore display saturation kinetics. The ability of this receptor to integrate the contributions of the various PCDDs and PCDFs in environmental mixtures may be compromised by the competitive nature of binding kinetics. AHH induction in mice simultaneously exposed to 2,3,7,8-T_4CDD and 2,3,7,8-T_4CDF was significantly depressed when compared to the level of AHH activity induced by 2,3,7,8-T_4CDD alone (Rizzardini *et al*., 1983). Clearly further research is required to validate these assays.

6.3 CONCLUSIONS AND RECOMMENDATIONS ON CURRENT LIMITATIONS AND FUTURE RESEARCH NEEDS

Limitations to Existing Data Base

1. Most studies of incinerators are not comprehensive enough to make useful correlations between PCDD/PCDF emissions and other factors. i.e. either fly ash or stack samples taken, either PCDD only or 2,3,7,8-T_4CDD only measured. Few studies present analysis of PCDD/PCDF as well as possible precursors.

2. Only Ontario studies have included analysis of feedstock as well as fly ash/stack emissions. This total analysis is needed to develop PCDD/PCDF mass balances in incinerators.

3. Most investigators in the field of incinerator emissions admit the difficulties in comparing data from different studies. Few have stated, however, that because of the great differences in incinerator design, feedstock composition, and operating conditions of municipal incinerators, there is no *a priori* reason to expect emissions from these sources to be comparable. Each incinerator must be evaluated separately to accurately assess emissions.

4. Seasonal variations in incinerator emissions have not been thoroughly evaluated.

5. Reproducibility studies for stack emission quantitative determination of PCDD/PCDF have not been performed.

6. More analytical standards, especially of the more toxic congeners, are needed. Many have now been synthesized in private or research laboratories, but are still not available from commercial sources. Use of standards is often not discussed in studies that present quantitative data.

7. Much of the work to date has ignored the PCDFs.

8. It is not clear in most cases how the reported emissions or concentrations of PCDD/PCDF relate to human health, especially in terms of bioavailability.

9. Many studies report data without giving key experimental details such as recoveries, cleanup methods, confirmation techniques used, etc.

10. Background levels are generally unknown; these should be determined since it is now apparent that low level PCDD/PCDF contamination is much more ubiquitous than previously suspected.

11. Precision and accuracy of analytical data are often not specified or even determined. Many more round-robin laboratory studies and standard exchanges are needed.

12. Studies of the biological response of biological test systems to most PCDDs and PCDFs are lacking.

Recommendations on Future Research Required

1. All 210 PCDD/PCDF compounds should be available individually from commercial laboratories in crystalline form or in certified correct concentration solution.

2. Additional ^{13}C-labelled standards are needed.

3. Increased quality control is needed for quantitative analysis.

4. Round-robin studies for samples such as incinerator fly ash are needed.

5. Reproducibility of the entire analytical process including sampling must be determined for incinerator emissions testing.

6. Future incinerator testing should include feed stock analysis.

7. Background and/or ambient environmental levels of PCDD/PCDF in air, water and soil should be determined directly.

8. Some work of Dow Chemical should be confirmed by others i.e. PCDD/PCDF formation in cigarettes and automobile exhaust. Further studies of PCDD/PCDF emissions from woodburning (especially from forest fires) and fossil fuel burning should be conducted.

9. Toxicity/carcinogenicity data are required for many more PCDD/PCDF compounds and the effect of mixtures of these substances should be investigated further. Specifically the relative non-toxic properties of O_8CDD should

be clarified in light of the predominance of this isomer in incinerator emissions, soils, sludges and our diet.

10. Methods of determining long-term human health effects from exposure to trace levels of chemicals are needed. The validity of extrapolating these effects from laboratory animal studies is questionable.

11. Studies attempting to relate PCDD/PCDF formation to incinerator conditions or levels of precursors will probably not be successful unless a truly comprehensive program is established in which hundreds of tests over a lengthy time (say, one year) are taken. Although it is unlikely that the results from such an investigation would justify the expense, some studies are needed in which a single incinerator is studied over a much longer time than has currently been reported. For example, if a link between fly ash concentrations of PCDD/PCDF and stack emissions could be established, then incinerator emissions could be monitored inexpensively by periodic analysis of fly ash. Such information cannot be obtained by monitoring an incinerator only 2 - 3 times. The accuracy of estimating annual emissions from only a few tests is questionable.

12. Methods of determining 2,3,7,8-T_4CDD toxic equivalents in vitro using cultured cells are needed to add biological realism to existing analytical methods. Such methods may be incorporated into future PCDD/PCDF standards.

13. Development of in situ biological methods to monitor impact points in Ontario are needed. Such methods could compare biological activities and PCDD/PCDF concentrations in locally restrained populations of animals or plants with control populations in unaffected areas.

14. In conjunction with Health and Welfare Canada MOE should undertake a food basket survey to quantify PCDD and PCDF residues in the diet.

15. Further monitoring of levels of PCDDs and PCDFs other than 2,3,7,8-T_4CDD in Ontario fish is required.

APPENDIX I

(Birmingham, Gehring, 1983)

ESTIMATION OF CURRENT LEVELS OF EXPOSURE FROM ADIPOSE TISSUE LEVELS IN HUMANS

1. Derivation of Equ. 1 in Table 5-26 (based on Casarett & Doull, 2nd ed. pp. 48-50).

$$\bar{X}_{ss} = C_{ss} \times V_d$$

Where $\bar{X}_{ss}$ = "Average" body burden at steady state,
C_{ss} = Plasma concentration at steady state and
V_d = Volume of distribution

Since $$\bar{C}_{ss} = \frac{F \times D}{V_d \times k_el \times t}$$

Where k_{el} = apparent first order rate constant

therefore $$\bar{X}_{ss} = \frac{F \times D \times V_d}{V_d \times k_{el} \times t} = \frac{F \times D}{k_{el} \times t}$$

Also $$k_{el} = 0.693/t^1/_2$$

Therefore $$\bar{X}_{ss} = \frac{F \times D \times t^1/_2}{0.693 \times t}$$

$$= \frac{1.443 \times F \times D \times t^1/_2}{t}$$

F = Fraction of chemical absorbed (Assume 100%)

D = Intravenous dose

Both F and D are related to the (Total daily dose) following exposure,

Therefore $$\bar{X}_{ss} = \frac{1.443 \times \text{Total daily dose} \times t^1/_2}{t}$$

Therefore Current level of Exposure
= Total daily dose = $$\frac{\bar{X}_{ss} \times t}{1.443 \times t^1/_2}$$

t = Time interval between exposure (daily).

t1/2 = half life (in days).

2. **Estimation of "Average" body burden (X_{ss}) at steady state.**

From section 4.3.4.5 a good background level for 2,3,7,8-T_4CDD toxic equivalents in human adipose tissue is 29 ppt.

a) **Assume:** 29 ppt for level of 2,3,7,8-T_4CDD toxic equivalents in adipose tissue

Therefore For lean 70 Kg man (with 10% fat)

$$7 \text{ kg} \times 29 \text{ ng/kg} = 203 \text{ ng in fat.}$$

b) **Assume:** concentration in liver is equal to that in adipose tissue

If wt. of liver = 1.5 kg.

Therefore $1.5 \text{ kg} \times 29 \text{ ng/kg} = 43.5 \text{ ng in liver.}$

c) **Assume:** concentration in rest of the body is 100 fold less than adipose tissue levels

Therefore $61.5 \text{ kg} \times 0.29 \text{ ng/kg} = 17.8 \text{ ng in body}$

Therefore: "Average" body burden $= \bar{X}_{ss} = \text{(Fat)} + \text{(Liver)} + \text{(Rest of the Body)}$

$$= 264.3 \text{ ng/70kg man.}$$

3. **Estimate current level of exposure**

Assume: $t = 1$ day (i.e., daily exposure)

$t^1/_2 = 365$ days

Therefore: Current level of Exposure = Total daily dose

$$= \frac{X_{ss} \times t}{1.443 \times t^1/_2}$$

$$= \frac{264.3 \text{ ng/70 kg} \times 1 \text{ day}}{1.443 \times 365 \text{ days}}$$

$$= 0.502 \text{ ng/70 kg/day}$$

$$= 502 \text{ pg/70 kg/day}$$

Therefore Current level of Exposure = 7.2 pg/kg.bw/day

TABLE 3.2.6B

BIO-AVAILABILITY RELATED STUDIES

SPECIES/ ROUTE	FORMULATION AND AMOUNT ADMINISTERED	DOSE	EXPOSURE TIME	NO. OF ANIMALS	ORGAN/QUANTITY	COMMENTS/CONCLUSIONS	REFERENCE
Rat/Oral	50% Ethanol	14.7 ng	24 hrs	7	Liver/36.7% ± 1.2% of dose	In going from fully available TCDD in ethanol to soil-adsorbed and activated carbon-adsorbed TCDD, availability of TCDD seems to be affected as shown by the decreasing concentration in the liver. Also increased contact time with the soil further reduces availability.	Poiger and Schlatter 1980
	Aqueous suspension of soil (37% w/w; 0.5 ml) that had been in contact with TCDD for: 10-15 hrs	12.7,22.9 ng	24 hrs	17	Liver/24.1% ± 4.8 % of dose		
	8 days	21.2,22.7 ng	24 hrs	10	Liver/16% ± 2.2% of dose		
	Aqueous suspension of activated carbon (25% w/w; 0.5 ml)	14.7 ng	24 hrs	6	Liver/0.07% of dose		
Rat/ Dermal	Methanol, 50 ul	26 ng		6	Liver/14.8% ± 2.6% of dose	Similar decreasing availability is shown in going from fully available TCDD in methanol to soil and activated carbon adsorbed TCDD.	Ibid
	Soil/water paste, 75 mg (corresponding to 50 mg of dry soil)	26 ng		5	Liver/0.05% of dose		
		350 ng		5	Liver/1.7% ± 0.5% of dose		
		1300 ng		3	Liver/2.2% ± 0.5% of dose		
	Activated carbon/water paste, 100 mg (corresponding to 50 mg of dry carbon)	26 ng		4	Liver/<0.05% of dose		
		1300 ng		4	Liver/<0.05% of dose		

TABLE 3.2.6B

BIO-AVAILABILITY RELATED STUDIES

SPECIES/ ROUTE	FORMULATION AND AMOUNT ADMINISTERED	DOSE	EXPOSURE TIME	NO. OF ANIMALS	ORGAN/QUANTITY	COMMENTS/CONCLUSIONS	REFERENCE
Rabbit/ Dermal (Ear Surface)	Acetone solution: (10-50 ul/animal)	0.6-1.5 ug/ ear	24 hrs			The minimum dose that induced lesions was 1 ug/ ear for fully available 2,3,7,8-T_4CDD rising to 160 ug/ear for activated carbon-adsorbed T_4CDD.	Ibid
	Soil/water paste (50 mg/animal)	2-3 ug/ear	24 hrs				
	Activated carbon/water paste (50 mg/animal)	160 ug/ear	24 hrs				
Rat/Oral	Crude fly ash extract (2 ml/24 hrs for 19 days)	Cumulative Dose			Average of duplicate experiment		
	T_4CDD Conc:322 ng/ml x (2x19)	12.2 ug	19 d		Liver/1.5 ng/g of liver		van der Berg, et al, 1983
	P_5CDD Conc:493 ng/ml x (2x19)	18.7 ug	"		Liver/17.8 ng/g of liver		
	H_6CDD Conc:500 ng/ml x (2x19)	19.0 ug	"		Liver/36.8 ng/g of liver		
	T_4CDF Conc:400 ng/ml x (2x19)	15.2 ug	"		Liver/8.1 ng/g of liver		
	P_5CDF Conc:485 ng/ml x (2x19)	18.4 ug	"		Liver/39.3 ng/g of liver		
	H_6CDF Conc:438 ng/ml x (2x19)	16.6 ug	"		Liver/64.2 ng/g of liver		

APPENDIX II

Page 3 of 3

TABLE 3.2.6B

BIO-AVAILABILITY RELATED STUDIES

SPECIES/ ROUTE	FORMULATION AND AMOUNT ADMINISTERED	DOSE	EXPOSURE TIME	NO. OF ANIMALS	ORGAN/QUANTITY	COMMENTS/CONCLUSIONS	REFERENCE
Rat/Oral	Purified fly ash extract (2 ml/24 hrs for 19 days)						Ibid
	T_4CDD Conc:475 ng/ml x (2x19)	18.1 ug	"		Liver/2.4 ng/g of liver		
	P_5CDD Conc:580 ng/ml x (2x19)	22.0 ug	"		Liver/20.1 ng/g of liver		
	H_6CDD Conc:539 ng/ml x (2x19)	20.5 ug	"		Liver/58.6 ng/g of liver		
	T_4CDF Conc:609 ng/ml x (2x19)	23.1 ug	"		Liver/21.0 ng/g of liver		
	P_5CDF Conc:624 ng/ml x (2x19)	23.7 ug	"		Liver/60.5 ng/g of liver		
	H_5CDF Conc:587 ng/ml x (2x19)	22.3 ug	"		Liver/102.4 ng/g of liver		
Rat/Oral	Fly Ash (2g/24 hrs.for 19 days						Ibid
	T_4CDD Conc:245 ng/g	9.3 ug	19 days		Liver/0.8 ng/g of liver		
	P_5CDD Conc:422 ng/g	16.0 ug	"		Liver/2.7 ng/g of liver		
	H_6CDD Conc:562 ng/g	21.3 ug	"		Liver 5.2 ng/g of liver		
	T_4CDF Conc:314 ng/g	11.9 ug	"		Liver 3.5 ng/g of liver		
	P_5CDF Conc:500 ng/g	19.0 ug	"		Liver 10.1 ng/g of liver		
	H_6CDF Conc:660 ng/g	25.1 ug	"		Liver/13.6 ng/g of liver		

TABLE 3.2.6C

BIO-AVAILABILITY RELATED STUDIES

SPECIES/ ROUTE	FORMULATION AND AMOUNT ADMINISTERED	EXPOSURE TIME	NO. OF ANIMALS	ORGAN/QUANTITY	COMMENTS/ CONCLUSIONS	REFERENCE
Rat/Oral	Fly ash extract/Total T_4CDD	19 days	4	Liver/0.16 ± 0.01% of dose		Ibid
	Fly ash extract/2,3,7,8-T_4CDD	19 days	4	Liver/3.7 ± 0.3% of dose		
	Fly ash/2,3,7,8-TCDD	19 days		Liver/0.9% of dose**		
	Fly ash extract/Total T_4CDF	19 days	4	Liver/0.89 ± 0.59% of dose		Ibid
	Fly ash extract/2,3,7,8-T_4CDF	19 days	4	Liver/1.0 ± 0.7% of dose		
	Fly ash/2,3,7,8-TCDF	19 days		Liver/0.3% of dose**		
	Fly ash extract/2,3,7,9-T_4CDF*	19 days	4	Liver/6.7 ± 5.7% of dose		
	Fly ash/2,3,7,9-TCDF*	19 days		Liver/1.7% of dose**		
	Fly ash extract/2,3,6,8-T_4CDF*	19 days	4	Liver/6.014% of dose		
	Fly ash/2,3,6,8-TCDF*	19 days		Liver/1.6% of dose**		
	Fly ash extract/Total P_5CDD	19 days	4	Liver/1.18 ± 0.14% of dose		Ibid
	Fly ash extract/1,2,3,7,8-P_5CDD*	19 days	4	Liver/27.0 ± 3.1% of dose		
	Fly ash/1,2,3,7,8-P_5CDD*	19 days		Liver/1.6% of dose**		
	Fly ash extract/Total P_5CDF	19 days	4	Liver/2.94 ± 0.01% of dose		Ibid
	Fly ash extract/2,3,4,7,8-P_5CDF*	19 days	4	Liver/25.6 ± 1.8% of dose		
	Fly ash/2,3,4,7,8-P_5CDF*	19 days		Liver/3.8% of dose **		

* These isomer identifications are preliminary since standards were not available.

** Average of 2 experiments

TABLE 3.2.6C

BIO-AVAILABILITY RELATED STUDIES

SPECIES/ ROUTE	FORMULATION AND AMOUNT ADMINISTERED	EXPOSURE TIME	NO. OF ANIMALS	ORGAN/QUANTITY	COMMENTS/ CONCLUSIONS	REFERENCE
Rat/Oral	Fly ash extract/Total H_6CDD	19 days	4	Liver/2.99 ± 1.31% of dose		Ibid
	Fly ash extract/1,2,3,6,7,8-H_6CDD* Fly ash/1,2,3,6,7,8-H_6CDD*	19 days 19 days	4	Liver/19.9 ± 8.2% of dose Liver/1.1% of dose**		
	Fly ash extract/1,2,3,7,8,9-H_6CDD* Fly ash/1,2,3,7,8,9-H_6CDD*	19 days 19 days	4	Liver/8.8 ± 4.1% of dose Liver/0.5% of dose**		
	Fly ash extract/Total H_6CDF	19 days	4	Liver/5.30 ± 1.9% of dose		
	Fly ash extract/2,3,4,6,7,8-H_6CDF* Fly ash/2,3,4,6,7,8-H_6CDF*	19 days 19 days	4	Liver/15.5 ± 5.7% of dose Liver/1.4% of dose**		

* These isomer identifications are preliminary since standards were not available.

** Average of 2 experiments

TABLE 3.2.6D

NTP BIO-AVAILABILITY STUDIES

SPECIES/ ROUTE	FORMULATION AND AMOUNT ADMINISTERED	DOSE	EXPOSURE TIME	NO. OF ANIMALS	ORGAN/QUANTITY	COMMENTS/CONCLUSIONS	REFERENCE
GUINEA PIG/ORAL (GAVAGE)	T_4CDD contaminated mixture soil/water (50:50) Soil levels of T_4CDD:0,770 880 ppb (Soil samples from Times Beach, MO)	T_4CDD/kg bw 0 ug 1.3 ug 3.8 ug 12.8 ug	One single dose then observed for 30 days		T_4CDD content of liver (ppb) <1.0 1.0 ± 0.1 34.3 ± 6.0	Toxic effects were discovered in the spleen, adrenal glands, bone marrow, testicles, and bladder. These effects were similar to those seen in guinea pigs exposed to oral doses of pure T_4CDD at the 1 and 3 ug/kg bw levels. Analysis of the T_4CDD content of the livers of the exposed guinea pigs revealed a clear dose-response relation.	McConnell et al, 1984
RAT/ORAL (GAVAGE)	T_4CDD contaminated soil Soil level:880 ppb	0 ug 0.04 ug 0.2 ug 1.1 ug 5.5 ug	One single dose then sacri-ficed 6 days later		AHH induction (nmoles/min/g) 0.6 ± 0.1 2.6 ± 0.2 3.9 ± 0.2 9.5 ± 0.5 24.2 ± 1.5	A 4-40 fold increase in hepatic enzyme activity (AHH induction) was observed, the 4-fold being observed at the lowest dose of 0.04 ug/kg bw. These effects were similar to those seen in rats exposed to pure T_4CDD in corn oil at similar doses.	Ibid

APPENDIX III

A POSSIBLE DIOXIN MAXIMUM ALLOWABLE DAILY INTAKE PARTITIONING ALGORITHM

1. Assumption for standard man

(mass = 70kg; external surface area = 2 m^2 (head and hand = 0.2 m^2);
daily drinking water intake = 1.5 L; daily food intake = 1.5 kg; daily inhalation volume = 20 m^3).

On an annual basis - head and hands (once/day) exposure = 73 m^2; water intake = 548 L; food intake = 548 Kg; inhalation = 7300 m^3,
therefore, 1 kg food = 1 L water = 0.14 m^2 skin = 13.3 m^3 air.

2. Calculation of Annual Dose based on maximum allowable daily exposure (from Appendix I)

10 pg/kg body wt/day = 700 pg/70 kg/day
= 255,500 pg/70 kg/year
= 255.5 ng/70 kg/year

N.B. 29 ppt (adipose tissue) = 264 ng/70 kg (assuming 10% adipose)
(from Appendix I)

3. Calculated daily exposure via Soil Air, Food and Water if Maximum Allowable Daily Intake Partitioned Equally

255,500 pg/4 = 63875 pg/year/exposure route:

Soil 63875 pg/73 m^2 = 875 pg/m^2 say 900 pg/m^2 = 900 pg/g of soil(a)
Water 63875 pg/548 L = 117 pg/L say 100 pg/L
Food 63875 pg/548 Kg = 117 pg/Kg say 100 pg/kg
Air 63875 pg/7300 m^3 = 8.75 pg/m^3 say 10 pg/m^3

converting to daily exposure:

Soil 900 pg/m^2(a) x 0.2 = 180/70 kg = 2.6 pg/kg/day
Water 100 pg/L x 1.5 = 150/70 kg = 2.14 pg/kg/day
Food 100 pg/Kg x 1.5 = 150/70 kg = 2.14 pg/kg/day
Air 10 pg/m^3 x 20.0 = 180/70 kg = 2.9 pg/kg/day

Total approximately = 10 pg/kg/day

or using maximum amounts (b)in each route of exposure:

Soil 3000 pg/m^2(a)x 0.2 = 600/70 kg = 8.6 pg/kg/day
Water 400 pg/L x 1.5 = 600/70 kg = 8.6 pg/kg/day
Food 400 pg/Kg x 1.5 = 600/70 kg = 8.6 pg/kg/day
Air 30 pg/m^3 x 20.0 = 600/70 kg = 8.6 pg/kg/day

Total approximately = 35 pg/kg/day

(a) The 900 pg/m^2 refers to m^2 of skin surface not soil. One gm was used as representative of the daily deposition pattern of soil on skin. An external body surface area of 1m^2 would also be representative of the largest proportion of the population. Thus 900 pg of PCDD + PCDF/m^2 of skin translates to 900 pg/g (ppt) of soil, if 1 gm of soil is spread over 1m^2 of skin. The Centre for Disease Control (Kimbrough, 1983) recently set 1 ppb (1000 pg/g) of 2,3,7,8-T_4CDD in soil as a safe level.

(b) e.g. for water: $\frac{255500}{548\ \text{L}}$ pg = 466 pg/L; say 400 pg/L

BIBLIOGRAPHY

Abbruzzi, R., Belvedere, G., Bianchi, R., Caccia, S., Cantoni, L., Castelli, M.G., Cerletti, C., Chiabrando, C., Coccia, P., Colturani, F., Facchinetti, T., Fanelli, R., Frigerio, A., Gerna, M., Lanzoni, J., Latini, R., Manara, L., Martelli, G., Negrini, P., Pantarott, C., Recciha, M., Riva, E., Rossi, E., Salmona, M., Siloni, M., Szumido, T., Vecchi, A., and M. Zanol. 1977. Identification and quantitative determination of 2,3,7,8-tetrachlorodibenzo-para-dioxin in animals from contaminated areas. Fourth International Symposium on Mass Spectrometry in Biochemistry and Medicine. Riva del Gorda, Italy. June 20-22.

Abernethy, D.J., Huband, J.C., Greenlee, W.F., and C.J. Boreiko. 1984. The effect of TCDD upon transformation, initiation and promotion of C3H/10T 1/2 cells. Abstract, 15th Annual EMS Meeting, p.147.

Ahling, B., and Lindskog. 1982. Emissions of chlorinated organic substances from combustion. In Chlorinated Dioxins and Related Compounds - Impact on the Environment, Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 215-225.

Ahling, B., Lindskog, A., Jansson, B., and G. Sundstrom. 1977. Formation of polychlorinated dibenzo-p-dioxins and dibenzofurans during combustion of a 2,4,5-T formulation. Chemosphere 6:461-468.

Akermark, B. 1978. Photodechlorination of haloaromatic compounds. In Dioxin: Toxicological and Chemical Aspects, Cattabeni, F., Cavallaro, A. and G. Galli (eds.). Spectrum Publications Inc., New York. pp. 191-193.

Albro, P.W., and B.J. Corbett. 1977. Extraction and clean-up of animal tissues for subsequent determination of mixtures of chlorinated dibenzo-p-dioxins and dibenzofurans. Chemosphere 7:381-385.

Allen, J.R., Barzotti, D.A., Van Miller, J.P, Abrahamson, L.J., and J.J. Lalich. 1977. Morphological changes in monkeys consuming a diet of five hundred parts-per-trillion of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Food Cosmet. Toxicol. 15:401-410.

Althaus, F.R., Lawrence, S.D., Sattler, G.D., Longfellow, D.G., and H.C. Pitot. 1982. Chemical quantification of unscheduled DNA synthesis in cultured hepatocytes as an assay for the rapid screening of potential chemical carcinogens. Cancer Research 42:3010-3015.

American Conference of Governmental Industrial Hygienists (ACGIH), 1982. The committee guidelines for classification of experimental animal carcinogens. In: Threshold Limit Values for Chemical Substances and Physical Agents in the Work Environment with Intended changes for 1982, ACGIH, Cincinnati. pp. 41-55.

Ames, B.N., Lee, F.D., and W.E. Durstom. 1973. An improved bacterial test system for the detection and classification of mutagens and carcinogens. Proc. Nat. Acad. Sci. U.S.A. 70(3):782-786.

APCD, 1981. Source testing studies of poly-chlorinated dibenzo-p-dioxins (PCDD) and dibenzofurans (PCDF). Source Measurement Section, Pollution Measurement Division and Organic Methods and Analysis Section, Analytical Services Division, Environment Canada. 45 pp.

Arstila, A.U., Reggiani, G., Sorvari, T.E., Raisanen, S., and H.K. Wipf. 1981. Elimination of 2,3,7,8,-tetrachlorodibenzo-p-dioxin in goat milk. Toxicology Letters 9:215-219.

Ashby, J. 1982. Screening chemicals for mutagenicity: practices and pitfalls. In: Mutagenicity: New Horizons in Genetic Toxicology. J.A. Heddle (ed.). Academic Press, New York. pp. 1-33.

Ashby, J. 1984. Personal Communications.

ASME, 1981. Study on state-of-the-art of dioxin from combustion sources. Am. Soc. Mec. Eng., New York, N.Y.

Axelson, O., Sudell, L., Andersson, K., Edling, C., and H. Kling. 1980. Herbicide exposure and tumour mortality. An updated epidemiologic investigation of Swedish railroad workers. Scand. J. Work Environ. Health 6:73-79.

Baker, D.J., and R.A.F. Matheson. 1981. Dioxins in the vicinity of two Atlantic region wood preserving operations. Environmental Protection Service, Atlantic Region, Dartmouth, Nova Scotia. EPS-5-AR-81-2, 69 pp.

Balasso, A., Fichtner, C., Frare, G., Leoni, A., Mauri, C., and S. Facchetti. 1983. An efficient method for the quantitative determination of 2,3,7,8-TCDD at ppt level in soil samples. *Chemosphere* 12:499-502.

Ballschmiter, K., Zoller, W., Scholz, Ch., and A. Nottrodt. 1983. Occurrence and absence of polychlorodibenzofurans and polychlorodibenzodioxins in fly ash from municipal incinerators. *Chemosphere* 12:585-594.

Bandiera, S., Sawyer, T., Romkes, M., Zmudska, B., Safe, L., Mason, G., Key, B., and S. Safe. 1984. Polychlorinated dibenzofurans (PCDFs): Effects of structure on binding to the 2,3,7,8-TCDD cytosolic receptor protein, AHH induction and toxicity. *Toxicology* 32:131-144.

Barnes, D.G. 1983. Assessing TCDD emissions from municipal waste combustors. *Chemosphere* 12:645-655.

Barnes, D.G. 1983. "Dioxin" production from combustion of biomass and waste. Presented at the Symposium on Energy from Biomass and Wastes VII, Lake Buena Vista, Florida, January 24-28, 1983.

Barsotti, D.A., Abrahamson, L.J. and J.R. Allen. 1979. Hormonal alterations in female Rhesus monkeys fed a diet containing 2,3,7,8-tetrachlorodibenzo-p-dioxin. *Bull. Environ. Contam. Toxicol.* 21:463-469.

Bastomsky, C.H. 1977. Enhanced thyroxine metabolism and high uptake goiters in rats after a single dose of 2,3,7,8-tetrachlorodibenzo-p-dioxin. *Endocrinology* 95:1150-1155.

Beatty, P.W., and R.A. Neal. 1976. Induction of DT-diaphorase by 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). *Biochem. Biophys. Res. Commun.* 68:197-204.

Beatty, P.W., Vaughn, W.K., and R.A. Neal. 1978. Effects of alteration of rat hepatic mixed-function oxidase (MFO) activity on the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). *Toxicol. Appl. Pharmacol.* 45:513-519.

Belli, G., Bressi, G., Calligarich, E., Cerlesi, S., and S.P. Ratti. 1982. Analysis of the TCDD-distribution as a function of the underground depth for data taken in 1977 and 1979 in Zone A at Seveso (Italy). In *Chlorinated Dioxins and Related Compounds - Impact on the Environment*, Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 137-153.

Benfenati, E., Gizzi, F., Reginato, R., Fanelli, R., Lodi, M, and R. Tagliaferri. 1983. Polychlorinated dibenzo-p-dioxins (PCDD) and polychlorinated dibenzofurans (PCDF) in emissions from an urban incinerator. 2. Correlation between concentration of micropollutants and combustion conditions. *Chemosphere* 12:1151-1157.

Bennett, B.G. 1983. Exposure of man to environmental PCBs - An exposure commitment assessment. *The Science of the Total Environment* 29:101-111.

Berry, B.N., DiGiovanni, J., Juchaw, M.R., Bracken, W.M., Gleason, G.L., and T.J. Slaga. 1978. Lack of tumor-promoting ability of certain environmental chemicals in a two-stage mouse skin tumorgenesis assay. *Res. Comm. Chem. Pathol. Pharmacol.* 20:101-108.

Bolton, L. 1978. Seveso dioxin: No solution in sight. *Chem. Eng.* 85:78.

Bonaccorsi, A., *et al.* *Arch. Toxicol.* (in press).

Botre, C., Memoli, A., and F. Alhaique. 1978. TCDD solubilization and photodecomposition in aqueous solutions. *Environ. Sci. Technol.* 12:335-336.

Bowes, G.W., Mulvihill, M.J., Simoneit, B.R.T., Burlingame, A.L., and R.W. Risebrough. 1975. Identification of chlorinated dibenzofurans in American polychlorinated biphenyls. *Nature* 256:305.

Bradlaw, J.A., and J.L. Casterline Jr. 1979. Induction of enzyme activity in cell culture: A rapid screen for detection of planar polychlorinated organic compounds. *J. Assoc. Offic. Anal. Chem.* 62:904-916.

Bronzetti, G., Bauer, C., Corsi, C., Del Carratare, R., Neeri, R., and M. Paoline. 1983. Mutagenicity study of TCDD and ashes from urban incinerators *in vitro* and *in vivo* using yeast D7 strain. *Chemosphere* 12:549-553.

Brookes, P., and F.J. de Serres. 1981. Overview of assay system performance. In Evaluation of Short-Term Tests for Carcinogens. F.J. de Serres and J. Ashby (eds.). Elsevier/North Holland, New York. pp. 96-111.

Bumb, R.R., Crummett, W.B., Cutie, S.S., Gledhill, J.R., Hummel, R.H., Kagel, R.O., Lamparski, L.L., Luoma, E.V., Miller, D.L., Nestrick, T.J., Shadoff, L.A., Stehl, R.H., and J.S. Woods. 1980. Trace chemistries of fire: A source of chlorinated dioxins. Science 210:385-390.

Bumbaco, M.J. 1983. Report on a stack sampling program to measure the emissions of selected trace organic compounds, particulates, heavy metals and HCl from the Royal Jubilee Hospital Incinerator, Victoria, B.C. Environment Canada, Environmental Protection Service.

Buser, H.R. 1979. Formation of polychorinated dibenzofurans (PCDFs) and dibenzo-p-dioxins (PCDDs) from the pyrolysis of chlorobenzenes. Chemosphere 8:415-424.

Buser, H.R., and H.P. Bosshardt. 1976. Determination of polychlorinated dibenzo-p-dioxins and dibenzofurans in commercial pentachlorophenols by combined gas chromatography-mass spectrometry. J. Assoc. Office Anal. Chem. 59:562-569.

Buser, H.R., and H.P. Bosshardt. 1978. Polychlorierte dibenzo-p-dioxine, dibenzofurane und benzole in der asche kommunaler und industrieller verbrennungsanlagen. Mitt. Geb. Lebensmittelunters Hyg. 69:191-199.

Buser, H.R., and C. Rappe. 1978. Identification of substitution patterns in polychlorinated dibenzo-p-dioxins (PCDDs) by mass spectrometry. Chemosphere 7:199-211.

Buser, H.R., and C. Rappe. 1979. Formation of polychlorinated dibenzofurans (PCDFs) from the pyrolysis of individual PCB isomers. Chemosphere 8:157-174.

Buser, H.R., and C. Rappe. 1980. High-resolution gas chromatography of the 22 tetrachlorodibenzo-p-dioxin isomers. Anal. Chem. 52:2257-2262.

Buser, H.R., Bosshardt, H.P., and C. Rappe. 1978a. Identification of polychlorinated dibenzo-p-dioxin isomers found in fly ash. Chemosphere 7:165-172.

Buser, H.R., Bosshardt, H.P., Rappe, C., and R. Lindahl. 1978b. Identification of polychlorinated dibenzofuran isomers in fly ash and PCB pyrolyses. Chemosphere 7:419-429.

Buu-Hoi, N.P. 1972. Organs as targets of "dioxin" (2,3,7,8-tetrachlorodibenzo-p-dioxin) intoxication. Naturwissenshaften 59:174-175.

Canadian Wildlife Service. 1980. Data on Herring Gulls.

Cantoni, L., Rizzardini, M., Belvedere G., Cantoni, R., Fanelli, R., and M. Salmona. 1981. Induction of mixed function oxidase by chronic treatment with 2,3,7,8-tetrachloro-dibenzo-p-dioxin in female rats. Toxicology. 21:159-167

Carter, C.D., Kimbrough, R.D., Liddle, J.A., Clene, R.E., Zack, M.M. Jr., Barthel, W.F., Koehler, R.E., and P.E. Phillips. 1975. Tetrachlorodibenzodioxin: An accidental poisoning episode in horse arenas. Science 188:738-740.

Cavallaro, A., Bandi, G., Invernizzi, G., Luciani, L., Mongini, E., and A. Gorni. 1980a. Sampling, occurrence and evaluation of PCDDs and PCDFs from incinerated solid urban waste. Chemosphere 9:611-621.

Cavallaro, A., Bartolozzi, G., Carreri, D., Bandi, G., Luciani, L., Villa, G., Gorni, A., and G. Invernizzi. 1980b. A method for the determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin at ppt levels in vegetables by high resolution gas chromatography and low resolution mass spectroscopy. Chemosphere 9:623-628.

Cavallaro, A., Luciani, L., Ceroni, G., Rocchi, I., Invernizzi, G., and A. Gorni. 1982. Summary of results of PCDD analysis from incinerator effluents. Chemosphere 11:859-868.

Cheung, M.O., Gilbert, E.F., and R.E. Peterson. 1981. Cardiovascular teratogenesis in chick embryos treated with 2,3,7,8-tetrachlorodibenzo-p-dioxin. In Toxicology of Halogenated and Hydrocarbons, Health and Ecological Effects. M. Khan and R. Stanton (eds.). Pergamon Press, New York. pp. 202-208.

Chiu, C., Thomas, R.S., Lockwood, J., Li, K., Halman, R., and R.C. Lao. 1983. Polychlorinated hydrocarbons from power plants, wood burning and municipal incinerators. Chemosphere 12:607-616.

Choudhry, G.G., and O. Hutzinger. 1982. Photochemical formation and degradation of polychlorinated dibenzo-furans and dibenzo-p-dioxins. Residue Reviews 84:113-161.

Choudhry, G.G., Olie, K., and O. Hutzinger. 1982. Mechanisms in the thermal formation of chlorinated compounds including polychlorinated dibenzo-p-dioxins. In Chlorinated Dioxins and Related Compounds - Impact on the Environment. Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 275-301.

Clark, D., Gauldie, J., Szewczuk, M., and G. Sweeney. 1981. Enhanced suppressor cell activity as a mechanism of immunosuppression by 2,3,7,8-tetrachorodibenzo-p-dioxin Proc. Soc. Exp. Biol. & Med. 168:290-299.

Clark, D., Sweeney G., Safe, S., Hancock, E., Kilburn, D., and J. Gauldie. 1983. Cellular and genetic basis for suppression of cytotoxic T-cell generation by haloaromatic hydrocarbons. Immunopharmacolocy 6:143-153.

Clement, R.E. 1984. Analytical data for analysis of Elmira soil samples for PCDD/PCDF. Memorandum to W. Jackman, Ministry of the Environment. 3pp.

Clement, R.E., and F.W. Karasek. 1982. Distribution of organic compounds adsorbed on size-fractionated municipal incinerator fly ash particles. J. Chromatogr. 234:395-405.

Clement, R.E., and H.M. Tosine. 1983. Ontario Ministry of Environment, unpublished data.

Clement, R.E., Tosine, H., and B. Ali. 1984. Levels of polychlorinated dibenzo-p-dioxins and dibenzofurans in wood burning stoves and fireplaces. Chemosphere (in press).

Clement, R.E., Tosine, H.M., Osborne, J., Ozvacic, V., and G. Wong. (In press). Levels of chlorinated organics in a municipal incinerator. In Chlorinated dioxins and dibenzofurans in the total environment - II. Keith, L., Choudhary, G., and C. Rappe (eds.).

Clement, R.E., Viau, A.C., and F.W. Karasek. 1983. Daily variations in composition of extractable organic compounds in fly ash from municipal waste incinerator. (submitted to) J. Environ. Anal. Chem.

Clive, D., McDuen, R., Spector, J.F.S., Piper, C., and I.K.H. Mavournin. 1983. Specific gene mutations in L5178Y cells in culture. A report of the Environmental Protection Agency Gene-Tox program. Mutation Research 115:225-251.

Coburn, J. 1983. Zenon Environmental Ltd., Burlington Ont., personal communication.

Cochrane, W.P., Singh, J., Miles, W., Wakeford, B., and J. Scott. 1981. Analysis of technical and formulated products of 2,4-D for the presence of PCDD. In Impact of Chlorinated Dioxins & Related Compounds on the Environment. Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 209.

Cockerham, L.E., and A.L. Young. 1982. The absence of hepatic cellular anomalies in TCDD-exposed beachmice - A field study. Environ. Tox. Chem. 1:299-308.

Cockerham, L.E., Young, A.L., and C.E. Thalken. 1980. Histopathological and ultrastructural studies of liver tissue from TCDD-exposed beachmice, Peromyscus polionotus. Technical Report FJSRL-TR-80-008. F.J. Seiler Research Laboratory, United States Air Force Academy, Colorado, 61p.

Cocucci, S., Digerolamo, F., Verderio, A., Covallaro, A., Colli, G., Gorni, A., Invernizzi, G., and L. Luciani. 1979. Absorption and translocation of tetrachlorodibenzo-p-dioxin by plants from polluted soil. Experientia 35:482-484.

Coggon, D., and E.D. Acheson. 1982. Do phenoxy herbicides cause cancer in man? Lancet 1(8280):1057-1059.

Cohen, G.M., Bracken, W.M., Lyer, R.P., Berry, D.L., Selkirk, J.K., and T.J. Slaga. 1979. Anticarcinogenic effects of 2,3,7,8-tetra-chlorodibenzo-p-dioxin on benzo(a)pyrene and 7,12-dimethylbenz(a)-anthracene tumor initiation and its reltionship to DNA binding. Canc. Res. 39:4027-4033

Commoner, B. 1976. Reliability of bacterial mutagenesis techniques to distinguish carcinogenic and non carcinogenic chemicals. U.S. Environmental Protection Agency publication EPA - 6001-76-02-06.

Commoner, B., and R.E. Scott. 1976. Accidental contamination of soil with dioxin in Missouri: Effects and countermeasures. Dioxin Information Project, Scientists' Institute for Public Information, New York, N. Y. 12 pp.

Consolidated Hearing Board in the matter of an application for Victoria Hospital energy from waste plant, Westminster Campus, London, Ontario, 1983. Witness statements of witnesses for the Ministry of the Environment, Exhibits No. 101(A) and 101(B).

Cooper, J.A. 1980. Environmental impact of residential wood combustion emissions and its implications. J. Air Poll. Contr. Assoc. 30:855-861.

Corbet, R. L., Muir, D.C.G., and G.R.B. Webster. 1983. Fate of 1,3,6,8-T_4CDD in an outdoor aquatic system. Chemosphere 12(4/5):523-527.

Courtney, K.D. 1976. Mouse teratology studies with chlorodibenzo-p-dioxins. Bull. Environ. Contam. Toxicol. 16:674-681.

Courtney, K.D., and J.A. Moore. 1971. Teratology studies with 2,4,5-T and 2,3,7,8-TCDD. Toxicol. Appl. Pharmacol. 20:396-403.

Crosby, D.G. 1978. Conquering the monster - The photochemical destruction of chlorodioxins. In Disposal and Decontamination of Pesticides, M.V. Kennedy (ed.). ACS Symposium Series 73. American Chemical Society, Washington, D. C.

Crosby, D.G., Moilanen, K.W., and A.S. Wong. 1973. Environmental generation and degradation of dibenzodioxins and dibenzofurans. Environ. Health Perspect. 5:259-266.

Crosby, D.G., and A.S. Wong. 1973. Photodecomposition of 2,4,5-trichlorophenoxyacetic acid (2,4,5-T) in water. J. Agric. Fd. Chem. 21:1052-1054.

Crosby, D.G., and A.S. Wong. 1977. Environmental degradation of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Science 195:1337-1338.

Crow, K. 1983. Significance of cutaneous lesions in the symptomology of exposure to dioxins and other chloracnegens. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R., Young, A., and A. Gray (eds.). Plenum Press, New York. pp. 605-612.

Crow, K. 1982. Cutaneous effects of chloracnegens. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Toronto. pp. 525-533.

Crummett, W.B. 1983. Status of analytical systems for determination of PCDDs and PCDFs. Chemosphere 12:429-446.

Crummett, W.B. 1981. Comments on origin of chlorinated dioxins. Science 213:1060.

Czeizel, E., and J. Kiraly. 1976. Chromosome examinations in works producing klorinol and buvinal. In The Development of a Pesticide as a Complex Scientific Task, L. Banki Medicina, Budapest. pp. 239-256.

Czuczwa, J.M., and R.A. Hites. 1983. Environmental fate of combustion generated polychlorinated dioxins and furans. In Dioxins in the Environment. Michigan State University, East Lansing, Michigan. (In press).

Czuczwa, J.M., and R.A. Hites. 1984. Environmental fate of combustion-generated polychlorinated dioxins and furans. Environ. Sci. Technol. 18:444-450.

Czuczwa, J.M., McVeety, B.D., and R.A. Hites. 1984. Polychlorinated dibenzo-p-dioxins and dibenzofurans in sediments from Siskiwit Lake, Isle Royale. Science 226:568-569.

Decad, G.M., Birnbaum, L.S., and H.B. Matthews. 1981. Distribution and excretion of 2,3,7,8-tetrachloro-dibenzofuran in C57BL/6J and DBA/2J mice. Toxicol. Appl. Pharmacol. 59:564-573.

Decad, G.M., Birnbaum, L.S. and H.B. Matthews. 1982. Disposition of 2,3,7,8-tetrachlorodibenzofuran in guinea pigs, rats and monkeys. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Toronto. pp. 307-315.

DeRoos, F.L., and A. Bjorseth. 1979. TCDD analysis of fly ash sample. Report prepared by Battelle Columbus Laboratories for the U.S. Environmental Protection Agency. June 15, 1979.

desRosiers, P.E. 1983. Remedial measures and disposal practices for wastes containing dioxins. Chemosphere 12:727-44.

Di Lernia R., Crimaudo, C., and G. Pacchetti. 1982. The study of X-rays and TCDD effects on satellite associations may suggest a simple model for application in environmental mutagenesis. Hum. Genet. 61(1):42-47.

DiDomenico, A., Silano, V., Viviano, G., and G. Zapponi. 1980a. Accidental release of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) at Seveso, Italy. VI. TCDD levels in atmospheric particles. Ecotoxicol. Environ. Safety 4:346-356.

DiDomenico, A., Silano, V., Viviano, G., and G. Zapponi. 1980b. Accidental release of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) at Seveso, Italy. V. Environmental persistence of TCDD in soil. Ecotoxicol. Environ. Safety 4:339-345.

DiDomenico, A., Silano, V., Viviano, G., and G. Zaponi. 1980c. Accidental relase of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) at Seveso, Italy. IV. Vertical distribution in soil. Ecotoxicol. Environ. Safety 4:327-338.

DiDomenico, A., Viviano, G., and G. Zapponi. 1982. Environmental persistence of 2,3,7,8-TCDD at Seveso. In Chlorinated Dioxins and Related Compounds: Impact on the Environment, Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 105-114.

DiGiovanni, J.A., Viaje, D.L., Berry, T.J., Slaga, T.J., and M.R. Juchaw. 1977. Tumour-initiating ability of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) and Aroclor 1254 in the two-stage system of mouse skin carcinogenesis. Bull. Environ. Contam. Toxicol. 18:552-557.

Dolphin, R.J., and F.W. Willmott. 1978. Separation of chlorinated dibenzo-p-dioxins from chlorinated congeners. J. Chromatogr. 149:161-168.

Dow Chemical. 1978. The trace chemistries of fire - a source of and routes for the entry of chlorinated dioxins into the environment. 46 pp.

ECETOC. 1982. Hepatocarcinogenesis in laboratory rodents: relevance to man. Monograph No. 4. European Chemical Industry Ecology and Toxicology Centre, Brussels, Belgium. 80 pp.

Eiceman, G.A., Clement, R.E., and F.W. Karasek. 1979. Analysis of fly ash from municipal incinerators for trace organic compounds. Anal. Chem. 51:2343-2350.

Eiceman, G.A., Clement, R.E., and F.W. Karasek. 1981. Variations in concentrations of organic compounds including polychlorinated dibenzo-p-dioxins and polynuclear aromatic hydrocarbons in fly ash from a municipal incinerator. Anal. Chem. 53:955-959.

Eiceman, G.A., Viau, A.C., and F.W. Karasek. 1980. Ultrasonic extraction of polychlorinated dibenzo-p-dioxins and other organic compounds from fly ash from municipal incinerators. Anal. Chem. 52:1492-1496.

ENVIROCON LTD. 1984. Report on combustion testing program at the SWARU plant, Hamilton-Wentworth. Prepared for Air Resources Branch, M.O.E. Report No. ARB-43-84-ETRD.

Environment Canada. 1983. Evidence of "S" area leaks found in the Niagara River. News Release, December 9, 1983.

Eriksson, M., Hardell, L., Berg, N.O., Moller, T., and O. Axelson. 1981. Soft tissue sarcomas and exposure to chemical substances: A case referent study. Br. J. Ind. Med. 38:27-33.

Esposito, M.P., Tiernan, T.O., and F.E. Dryden. 1980. Dioxins. U. S. Environmental Protection Agency. EPA-600-80-197. 351 pp.

Esposito, M.P., and D.R. Watkins. 1980. Airborne dioxins: The problem in review. Paper 80-10.1 at 73rd. Annual Meeting Air Pollution Control Association, Montreal, Quebec. June 22-27.

Facchetti, S., Formari, A., and M. Montagna. 1980. Distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the tissues of a person exposed to the toxic cloud at Seveso (Italy). Adv. Mass. Spectrom. 83:1405-1414.

Fanelli, R., Bertoni, M.P., Castelli, M.E., Chiabrando, C., Martelli, G.P., Noseda, A., Garattini, S., Binaghi, C., Marazza, V., and F. Pezza. 1980a. 2,3,7,8-tetrachlorodibenzo-p-dioxin toxic effects and tissue levels in animals from the contaminated area of Seveso, Italy. Arch. Environm. Contam. Toxicol. 9:569-577.

Fanelli, R., Bertoni, M.P., Bonfonti, M., Castelli, M.G., Chiabrando, C., Martelli, G.P., Noe, M.A., Noseda, A., Garrattini, S., Binaghi, C., Morazza, V., Pezza, F., Pozzoli, D., and G. Ocognetti. 1980b. 2,3,7,8-tetrachlorodibenzo-p-dioxin levels in cow's milk from the contaminated area of Seveso, Italy. Bull. Environ. Contam. Toxicol. 24:634-639.

Fara, G., Del Corono, G., Bonetti, F., Caramaschi, F., Dardanoni, L., Favaretti, C., Giambelluca, S., Marni, E., Mocarelli, P., Montesarchio, E., Puccinelli, V., and C. Volputo. Chloracne after release of TCDD at Seveso, Italy. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Toronto. pp. 545-559.

Firestone, D., Ress, J., Brown, N.L., Barron, R.P., and J.N. Damico. 1972. Determination of polychlorinated dibenzo-p-dioxins and related compounds in commercial chlorophenols. J. Assoc. Offic. Anal. Chem. 55:85-92.

Fishbein, L. 1974. Mutagens and potential mutagens in the biosphere. I: DDT and its metabolites, PCBs, Chlorodioxins, PAH, Haloethers, The Science of the Total Environment 2:305-340.

Flick, D.F., Firestone, D., and G.H. Higginbottam. 1972. Studies of the chick edema disease: 9. Response of chicks fed on singly administered synthetic edema-producing compounds. Poult. Sci. 51:2026-2034.

Forrette J.E., and A.L. Rozek. 1977. Analysis of technical dicamba for 2,7-dichlorodibenzo-p-dioxin and 2,3,7,8-tetrachlorodibenzo-p-dioxin. Proc. 11th Int. Velsicop Symp. pp. 1-6.

Fries, G.F. 1975. Retention and excretion of 2,3,7,8-tetrachlorodibenzo-p-dioxin by rats. J. Agric. Fd. Chem. 23:265-269.

Frigerio, A. 1978. The Seveso case: TCDD in animals from contaminated areas. Methodol. Surv. Biochem. 7:161-166.

Garattini, S., Vecchi A., Sironi, M., and A. Mantovani. 1982 immunosuppresant activity of TCDD in mice. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Frei, R., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Toronto. pp. 403-409.

Gasiewicz, T.A., and R.A. Neal. 1979. 2,3,7,8-tetrachlorodibenzo-p-dioxin: tissue distribution, excretion, and effects on clinical chemical parameters in guinea pigs. Toxicol. Appl. Pharmacol. 51:329-339.

Gehring, P., 1983. Personal communication.

Geiger, L.E., and R.A. Neal. 1981. Mutagenesis testing of 2,3,7,8-Tetrachlorodibenzo-p-dioxin in histidine auxotrophs of Salmonella typhimurium. Toxicol. Appl. Pharmacol. 59(1):125-129.

Getzendaner, M.D., Mahle, N.H., and H.S. Higgins. 1977. Search for the presence of 2,3,7,8-tetrachlorodibenzo-p-dioxin in bovine milk. Bull. Environ. Contam. Toxicol. 18:2.

Gierthy, J.F., and D. Crane. 1983. In vitro bioassay for chlorinated dioxins and dibenzofurans. American Chemical Society, Division of Environmental Chemistry, 186th National Meeting, Washington, D.C. Preprint of Abstracts 23:163-166.

Gilbertson, M., and A. Fox. 1983. Chick edema disease and hepatic porphyria in Lake Ontario herring gull embryos in the early 1970s. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L., and A.P. Gray (eds.). Plenum Press, New York. pp. 341-356.

Gilbertson, M. 1982. Etiology of chick edema disease in herring gulls in the lower Great Lakes. Canadian Technical Report of Fisheries and Aquatic Sciences 1120. Fish Habitat Management Branch, Department of Fisheries and Oceans. Ottawa, Ontario. pp. 1-14.

Gilbert, P.G., Saint-Ruf, G., Poncelet, F., and M. Mercier. 1980. Genetic effects of chlorinated anilines and azo benzenes on Salmonella typhimurium. Arch. Environ. Contam. Toxicol. 9(5):533-541.

Gizzi, F., Reginato, R., Benfenati, E., and R. Fanelli. 1982. Polychlorinated dibenzo-p-dioxins (PCDD) and polychlorinated dibenzofurans in emissions from an urban incinerator. 1. Average and Peak Values. Chemosphere 11:577-583.

Graham, M., Hileman, F., Kirk, D., Wendling, J., and J. Wilson. 1984. Distribution of dioxins and furans in human autopsy tissues from the general population. 4th International Symposium on Chlorinated Dioxins and Related Compounds. Ottawa, October 16 - 18, 1984, Poster 36.

Grant, D.L. 1977. In: Proceedings of the 12th Annual Workshop in Pesticide Residue Analysis (held in Winnipeg). pp. 251.

Green, S., and F.S. Moreland. 1975. Cytogenetic evaluation of several dioxins in the rat. Toxicol. Appl. Pharmacol. 33:161

Green, S., Moreland, F.S., and C. Sheu. 1977. Cytogenetic effect of 2,3,7,8-tetrachlorodibenzo-p-dioxin on rat bone marrow cells. U.S. FDA, Washington D.C., FDA By-Lines 6:292.

Greig, J.B., Jones, G., Butler, W.H., and J.M. Barnes. 1973. Toxic effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Food Cosmet. Toxicol. 11:585-595.

Greig, J.B., and G. Osborne. 1978. Changes in rat hepatic cell membranes during 2,3,7,8-tetrachloro-dibenzo-p-dioxin intoxication. In Dioxin Toxicological and Chemical Aspects. Cattabeni, F., Cavallaro, A., and G. Galli (eds.). S.P. Medical & Scientific Books, New York, London. pp. 105-111.

Gross, M.L., Weerasinghe, N.C.A., Meehan, J.L., and R. Harless. 1983. The analysis of tetrachlorodibenzo-p-dioxins and tetrachlorodibenzofurans in chemical waste and in the emission from its combustion. American Chemical Society, Division of Environmental Chemistry, 186th National Meeting, Washington, D. C., Preprint Extended Abstr. 23:256-259.

Guo, P.H.M., Fowlie, P.J.A., Cairns, V.W., and B.E. Jank. 1979. Activated sludge treatment of a wood preserving effluent containing PCP. Environment Canada, EPS Report (Draft).

Gupta, B.N., Vos, J.G., Moore, J.A., Zinkl, J.G., and B.C. Bullock. 1973. Pathologic effects of 2,3,7,8-tetrachloro-dibenzo-p-dioxin in laboratory animals. Environ. Health Perspect. 5:125-140.

Haile, C.L., Stanley, J.S., Magin, A.M., Northcott, R.V., and D.P. Redford. 1982. Emissions of organic pollutants from coal-fired utility border plants. Joint Report. Midwest Research Institute, Kansas City and Office of Pesticides and Toxic Substances, USEPA Washington. pp. 1-22.

Hall, R.E., and D.E. DeAngelis. 1980. EPA's research program for controlling residential wood combustion emissions. J. Air Poll. Contr. Assoc. 30:862-867.

Hardell, L. 1981. Relation of soft tissue sarcoma, malignant lymphoma and colon cancer to phenoxy acids, chlorophenols and other agents. Scand. J. Work. Environ. Health 7:119-130.

Hardell, L., and A. Sandstrom. 1978. Malignant mesenchymal soft tissue tumors and exposure to phenoxy acids or chlorophenols (Sweden). Lakartidningen 75:3535-3536.

Hardell, L., and A. Sandstrom. 1979. Case control study: Soft tissue sarcomas and exposure to phenoxyacetic acids or chlorophenols. Br. J. Cancer 39:711-717.

Hardell, L., Eriksson, M., Lenner, P., and E. Lundgren. 1981. Malignant lymphoma and exposure to chemicals, especially organic solvents, chlorophenols, and phenoxy acids: A case control study. Br. J. Cancer 43:169-176.

Harless, R.L., and R.G. Lewis. 1982. Quantitative determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin residues by gas chromatography/mass spectrometry. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R.W., Merian, E. and F. Pocchiari(eds.). Pergamon Press, Oxford. pp. 25-35.

Harless, R.L., Oswald, E.O., Lewis, R.G., Dupuy, A.E., and D.D. McDaniel. 1982. Determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin in fresh water fish. Chemosphere 11(2):193-198.

Harless, R.L., Oswald, E.O., Wilkinson, M.K., Dupuy, A.E. Jr., McDaniel, D.D., and Han Tai. 1980. Sample preparation and gas chromatography-mass spectrometry determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Anal. Chem. 52:1239-1245.

Hassoun, E.M., and L. Dencker. 1982. TCDD embryotoxicity in the mouse may be enhanced by B-naphthoflavone, another ligand of A_h-receptor. Toxicol. Lett. 12:191-198.

Hass, J.R., and M.D. Friesen. 1979. Qualitative and quantitative methods for dioxin analysis. Annals N.Y. Acad. Sci. 320:28-42.

Hass, J.R., Friesen, M.D., Harvan, D.J. and C.E. Parker. 1978. Determination of polychlorinated dibenzo-p-dioxins in biological samples by negative chemical ionization mass spectrometry. Anal. Chem. 50:1474-1479.

Hawkes, C.L., and L.A. Norris. 1977. Chronic oral toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) to rainbow trout. Trans. Amer. Fish Society 106(6):641-645.

Hay, A. 1980. Exposure to TCDD: The health risks. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Toronto. pp. 589-600.

Hay, A., Ashby, J., Styles, J.A., and B. Elliot. 1983. The mutagenic properties of 2,3,7,8-tetrachlorodibenzo-p-dioxin. American Chemical Society, Division of Environmental Chemistry, 186th National Meeting, Washington D.C., Preprinted Extended Abstract 23(2):14-22.

Health and Welfare Canada and Environment Canada. 1983. Report of the Joint Health and Welfare Canada/ Environment Canada Expert Advisory Committee on Dioxins. 57 pages.

Helder, Theo. 1980a. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on early life stages of the pike (Esox lucius L.). The Science of the Total Environment 14:255-264.

Helder, Theo. 1980b. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on early life stages of two freshwater fish species. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R., Meriam, E., and F. Pocchiari (eds.). Pergamon Press, Toronto. pp. 455-462.

Helder, Theo. 1981. Effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on early life stages of rainbow trout (*Salmo gairdneri*, Richardson). *Toxicology* 19:101-112.

Helling, C.S., Isensee, A.R., Woolson, E.A., Ensor, P.D.J., Jones, G.E., Plimmer, J.R., and P.C. Kearney. 1973. Chlorodioxins in pesticides, soils and plants. *J. Environ. Quality* 2:171-178.

Henck, J.W., Newn, M.A., Kociba, R.J., and K.S. Rao. 1981. 2,3,7,8-tetrachlorodibenzo-p-dioxin: Acute oral toxicity in hamsters. *Toxicol. Appl. Pharmacol.* 59:405-407.

Highman, B., Gaines, T.B., and H.J. Schumacher. 1976. Retarded development of fetal renal alkaline phosphatase in mice given 2,4,5-trichlorophenoxyacetic acid. *J. Toxicol. Environ. Health* 2:1007-1018.

Hobson, L.B., Lee, L.E., Gross, M.L., and A.L. Young. 1983. Dioxin in body fat and health status: A feasibility study. American Chemical Society, Division of Environmental Chemistry, 186th National Meeting, Washington, D.C. Preprint of Abstracts 23:91-93.

Hoering, A. June, 1983. Personal Communication to A. Johnson.

Hogstedt, C., and B. Westerlund. 1980. Cohort study of the causes of death among forestry workers with and without exposure to phenoxy acid preparations (in Swedish). *L kartidningen* 77:1828-1831.

Homberger, E., Reggiani, E., Sambeth, J., and H.K. Wipf. 1979. The Seveso accident: Its nature, extent, and consequences. *Ann. Occup. Hyg.* 22:327-370.

Hryhorczuk, D.O., Withrow, W.A., Hesse, C.S., and V.R. Beasley. 1981. A wire reclamation incinerator as a source of environmental contamination with tetrachlorodibenzo-p-dioxins and tetrachlorodibenzofurans. *Arch. Environ. Health* 36:228-234.

Hummel, R.A. 1977. Clean-up techniques for the determination of parts-per-trillion residue levels of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). *J. Agric. Fd. Chem.* 25:1049-1053.

Hummel, R.A., and L.A. Shadoff. 1980. Specificity of low resolution gas chromatography-low resolution mass spectrometry for the detection of tetrachlorodibenzo-p-dioxin in environmental samples. Anal. Chem. 52:191-192.

Hussain, S., Ehrenberg, L.E., Lafroth, G., and T. Gejvall. 1972. Mutagenic effects of TCDD on bacterial systems. Ambio. 1:32-33.

Hutzinger, O., Olie, K., Lustenhouwer, J.W.A., Okey, A.B., Bandiera, S., and S. Safe. 1981. Polychlorinated dibenzo-p-dioxins and dibenzofurans: A bioanalytical approach. Chemosphere 10:19-25.

Hutzinger, O., Safe, S., Wentzel, B.R., and V. Zitko. 1973. Photochemical degradation of di- and octachlorodibenzofuran. Environ. Health Perspect. 5:267-271.

ICRP (International Commission on Radiation Processes). 1968. Report of Committee IV, Publication No. 10.

Interdepartmental Committee on Toxic Chemicals, 1983. Dioxins in Canada: The Federal Approach. Environment Canada, Hull, Quebec, K1A 1C8. 39 pages.

International Agency for Research on Cancer (IARC). 1983a. Monographs on the evaluation of the carcinogenic risk of chemicals to humans. Miscellaneous Pesticides 30:37-56.

International Agency for Research on Cancer (IARC). 1983b. IARC monograph programme on the evaluation of the carcinogenic risk of chemicals to humans - preamble. 30:11-31.

Ioannou, Y.M., Birnbaum, L.S., and H.B. Matthews. 1983. Toxicity and distribution of 2,3,7,8-tetrachlorodibenzofuran in male guinea pigs. J. Toxicol. Environ. Health 12:541-553.

Isensee, A.R. 1978. Bioaccumulation of 2,3,7,8-tetrachlorodibenzo-para-dioxin. In Chlorinated Phenoxy Acids and Their Dioxins, C. Ramel (ed.). Ecological Bulletin No. 27. Swedish Natural Science Research Council. pp. 255-262.

Isensee, Alan R., and G.E. Jones. 1975. Distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in aquatic model ecosystems. Environ. Sci. Tech. 9(7):668-6721.

Jackson, W.T. 1972. Regulation of mitosis. III. Cytological effects of 2,4,5,-T and of dioxin contaminants in 2,4,5,-T formulations. J. Cell Sci. 10:15-25.

Jansson, B., and G. Sundstrom. 1982. Formation of polychlorinated dibenzofurans (PCDF) during a fire accident in capacitors containing polychlorinated biphenyls (PCBs). In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Huntzinger, O., Frei, R.W., Merian, E., and F. Pocchiari, (eds.). Pergamon Press, Oxford. pp. 201-208.

Jansson, B., Sundstrom, G., and B. Ahling. 1978. Formation of polychlorinated dibenzo-p-dioxins during combustion of chlorophenol formulations. The Science of the Total Environment 10:209-217.

Jensen, D.J., Hummel, R.A., Mohle, N.H., Kocha, C.W., and H.S. Higgins. 1981. A residue study on beef cattle consuming 2,3,7,8-tetrachlorodibenzo-p-dioxin. J. Agri. Food Chem. 29:265-268.

Jensen, D.J., and R.A. Hummel. 1982. Secretion of TCDD in milk and cream following the feeding of TCDD to lactating dairy cows. Bull. Environ. Contam. Toxicol. 29:440-446.

Jensen, D.J., Getzendaner, M.E., Hummel, R.A., and J. Turley. 1983. Residue studies for (2,4,5-trichlorophenoxy) acetic acid and 2,3,7,8-tetrachlorodibenzo-p-dioxin in grass and rice. J. Agric. Fd. Chem. 31:118-122.

Jones, P.A. 1981. Chlorophenols and their impurities in the Canadian environment. EPS, Environment Canada, EPS 3-EC-81-2 (1981).

Kaczmar, S.W., Zabik, M.J., and F.M. D'Itri. 1983. Parts per trillion residues of 2,3,7,8-TCDD in Michigan fish. Report of O'Brien & Gere Engineers Inc., Syracuse, N.Y.

Kaminsky, L., DeCaprio, A.P., Gierthy, J.F., and J.B. Silkworth. 1984. The role of environmental matrices in chlorinated dioxin and dibenzofuran toxicity. 4th International Symposum on Chlorinated Dioxins and Related Substances. Ottawa, October 16 - 18, 1984.

Karasek, F.W. 1980. Dioxins from garbage: Previously unknown source of toxic compounds is neing uncovered using advanced analytical instrumentation. Can. Research (Sept.) 50-56.

Karasek, F.W., Clement, R.E., and A.C. Viau. 1982. Distribution of PCDD and other toxic compounds generated on fly ash particulates in municipal incinerators. J. Chromatogr. 239:173-180.

Karasek, F.W., and F.I. Onuska. 1982. Trace analysis of the dioxins. Anal. Chem. 54:309A.

Karasek, F.W., and A.C. Viau. 1983. Gas chromatographic - mass spectrometric analysis of polychlorinated dibenzo-p-dioxins and organic compounds in high-temperature fly ash from municipal waste incineration. J. Chromatogr. 265:79-88.

Kearney, P.C., Woolson, E.A., Isensee, A.R., and C.S. Helling. 1973. Tetrachlorodibenzodioxin in the environment: Sources, fate and decontamination. Environ. Health Perspec. 5:273-277.

Kemp, C.C. 1983. Notes on polychloro dibenzo dioxins and polychloro dibenzo furans in connection with waste-to-energy plants. Report to Browning-Ferris Industries, Inc., 74 pp.

Kenaga, E.E. 1975. The evaluation of the safety of 2,4,5-T to birds in areas treated for vegetation control. Residue Rev. 59:1-19.

Kenaga, E.E., and L.A. Norris. 1983. Environmental toxicity of TCDD. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L., and A.P. Gray (eds.). Plenum Press, New York. pp. 277-299.

Khera, K.S., and J.A. Ruddick. 1973. Polychlorodibenzo-p-dioxins: Perinatal effects and the dominant lethal test in Wistar rats. In Chlorodioxins - Origins and Fate. E.H. Blair (ed.). Adv. Chem. Ser. No. 120 Am. Chem. Soc., Washington, D.C. pp. 70-84.

Kimble, B.J., and M.L. Gross. 1980. Tetrachlorodibenzo-p-dioxin quantitation in stack-collected coal fly ash. Science 207:59-61.

Kimbrough, R.D., Falk, H., Stehr, P., and G. Fries. 1984. Health implications of 2,3,7,8- tetrachlorodibenzodioxin (TCDD) contamination of residential soil. J. Toxicol. Environ. Health 14:47-93.

Kimbrough, R.D., Carter, C.D., Liddle, L.A., and R.E. Clene. 1977. Epidemiology and pathology of a tetrachlorodibenzo-dioxin poisoning episode. Arch. Environ. Health 32(2):77-86.

King, F.G., Dedrick, R.L., Colins, J.M., Matthews, H.B., and L.S. Birnbaum. 1983. Physiological model for the pharmaco-kinetics of 2,3,7,8-tetrachlorodibenzofuran in several species. Toxicol. Appl. Pharmacol. 67:390-400.

Kitchin, K.T., and J.S. Woods. 1978. 2,3,7,8-tetrachlorodibenzo-p-dioxin induction of aryl hydrocarbon hydroxylase (AHH) in hepatic microsomes from female rats. Toxicol. Appl. Pharmacol. 45(1):297.

Kleopfer, R.D., Bunn, W.W., Yue, K.T., and D.J. Harris. 1983. Occurrence of tetrachlorodibenzo-p-dioxin in environmental samples from southwest Missouri. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, G., Keith, L.H., and Rappe, C. (eds.). Butterworth Publishers, Boston. pp. 193-201.

Knutson, J.C., and A. Poland. 1980(a). 2,3,7,8-tetrachloro- dibenzo-p-dioxin: Failure to demonstrate toxicity in twenty-three cultured cell types. Toxicol. Appl. Pharmacol. 54:377-383.

Knutson, J.C., and A. Poland. 1980(b). Keratinization of mouse teratoma cell line XB produced by 2,3,7,8-tetrachlorodibenzo-p-dioxin: An in vitro model of toxicity. Cell 22:27-36.

Kociba, R.J., and B.A. Schwetz. 1982a. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Drug Metabolism Reviews 13:387-406.

Kociba, R.J., and B.A. Schwetz. 1982b. A review of the toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) with a comparison of the toxicity of other chlorinated dioxin isomers. Assoc. Food Drug Officials Quart. Bull. 46:168-188.

Kociba, R.J., Keeler, P.A., Park, C.N., and P.J. Gehring. 1976. 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD): Results of a 13-week oral toxicity study in rats. Toxicol. Appl. Pharmacol. 35:553-574.

Kociba, R.J., Keyes, D.G., Beyer, J.E., Carreson, R.M., Wade, E.E., Dittenber, D.A., Kalmins, P.O., Franson, L.F., Park, P.N., Barnard, S.D., Hummel, P.A., and C.G. Humiston. 1978. Results of a two year chronic toxicity and oncogenicity study of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in rats. Toxicol. Appl. Pharmacol. 46:279-303.

Kondorosi, A., Fedorcsak, I., Solymosy, F., Ehrenberg, L., and S. Osterman-Golkar. 1973. Inactivation of Q.B. RNA by electrophiles. Mutat. Res. 17:149-161.

Kooke, R.M.M., Lustenhouwer, J.W.A., Olie, K., and O. Hutzinger. 1981. Extraction efficiencies of polychlorinated dibenzo-p-dioxins and polychlorinated dibenzofurans from fly ash. Anal. Chem. 53:461-463.

Koshakji, R.P., Harbison, R.D., and M.T. Bush. 1984. Studies on the metabolic fate of [^{14}C] 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in the mouse. Toxicol. Appl. Pharmacol. 73:69-77.

Kouri, R.E., Rude, T.H., Joglekar, R., Dansette, P.M., Jerina, D.M., Atlas, S.A., Owens, I.S., and D.W. Nebert. 1978. 2,3,7,8-Tetrachlorodibenzo-p-dioxin as cocarcinogen causing 3-methylcholanthrene-initiated subcutaneous tumors in mice genetically "nonresponsive" at Ah locus. Cancer Res. 30:2777-2783.

Krystynak, R. 1983. An economic assessment of 2,4,-D in Canada: The case of grain. Can. Farm Economics 18(1):7.

Kunita N., Kashimoto, T., Miyata, H., Fukushima, S., Hori, S., and H. Obara. 1984. Causal agents of Yusho. Amer. J. Ind. Med. 5:45-58.

Kuroki, H., Haraquchi, K., and M. Masuda. 1984. Synthesis of polychlorinated dibenzofuran isomers and their gas chromatographic profiles. Chemosphere 13:561-573.

Kuroki, H., and Y. Masuda. 1978. Determination of polychlorinated dibenzofuran isomers retained in patients with Yusho. Chemosphere 7:771-777.

Lamberton, J., Griffin, D., Arbogast, B., Inman, R., and M. Deinzer. 1979. The determination of polychlorodibenzo-p-dioxins in pentachlorophenol and wood treatment solutions. Am. Ind. Hyg. Assoc. J. 40(9):816-822.

Lamparski, L.L., and T.J. Nestrick. 1980. Determination of tetra-, hexa-, hepta-, and octachlorodibenzo-p-dioxin isomers in particulate samples at parts per trillion levels. Anal. Chem. 52:2045-2054.

Lamparski, L.L., T.J. Nestrick, and S.S. Cutie. 1982. The impact of the environment on airborne particulate matter from the eruption of Mount Saint Helens. In Press.

Lamparski, L.L., Nestrick, T.J., and V.A. Stenger. 1984. Presence of chlorodibenzodioxins in a sealed 1933 sample of dried municipal sewage sludge. Chemosphere 13:361-365.

Langhorst, M.L., and L.A. Shadoff. 1980. Determination of parts-per-trillion concentrations of tetra-, hexa-, hepta- and octachlorodibenzo-p-dioxins in human milk samples. Anal. Chem. 52:2037-2044.

Leung, P.K., and A.R. Meadows. 1984. Identification of organic compounds emanating from plasma pyrolysis of polychlorinated biphenyls at the Royal Military College of Canada, Kingston, Ontario: June 8, 1983. Report from Source Measurement Section, Technical Services Branch, Environmental Protection Programs Directorate, Environment Canada. 22pp.

Levin, J.O., Rappe, C., and C.A.J. Nilsson. 1976. Use of chlorophenols as fungicides in sawmills. Scand. J. Work Environ. Health 2(2):71-81.

Liberti, A., and D. Brocco. 1982. Formation of polychlorodibenzodioxins and polychlorodibenzofurans in urban incinerators emissions. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R.W., Merian, E., and Pocchiari, F. (eds.). Pergamon Press, Oxford, pp. 245-251.

Liberti, A., Goretti, G., and M.V. Russo. 1983. PCDD and PCDF formation in the combustion of vegetable wastes. Chemosphere 12:661-663.

Liberti, A., Ciccioli, P., Brancaleoni, E., and A. Cecinato. 1982. Determination of polychlorodibenzo-p-dioxins and polychlorodibenzofurans in environmental samples by gas chromatography - mass spectometry. J. Chromatogr. 242:111-118.

Liberti, A., Brocco, D., and V. DiPalo. 1980. Evaluation of organochlorine compounds in the emissions of urban incinerators. In *Analytical Techniques in Environmental Chemistry*, J. Albaiges'(ed.). Pergamon Press. pp. 157-166.

Libich, S., To, J.C., Frank, R., and G.J. Sirons. 1984. Occupational exposure of herbicide applicators to herbicides used along electric power transmission line right-of-way. *Am. Ind. Hyg. Assoc. J.* 45:56-62.

Little, A.D. 1981. *Study on state-of-the-art of dioxin from combustion sources*. American Society of Mechanical Engineers. New York, N.Y. pp. 77.

Long, J.R., and D.J. Hanson. 1983. Dioxin issue focuses on three major controversies in U.S. *Chem. Eng. News* 61:30-32.

Loprieno, N., Sbrana, I., Rusciano, D., Lascialfari, D., and T. Lari. 1982. *In vivo* cytogenetics studies on mice and rats exposed to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). In *Chlorinated Dioxins and Related Compounds - Impact on the Environment*. Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 419-428.

Lowrance, W.W., (ed.). 1984. *Public Health Risks of the Dioxins*. The Rockefeller University, New York, 389 pages.

Lustenhouwer, J.W.A., Olie, K., and O. Hutzinger. 1980. Chlorinated dibenzo-p-dioxins and related compounds in incinerator effluents: A review of measurements and mechanisms of formation. *Chemosphere* 9:501-522.

Macek, K.J., Petrocelli, S.R., and B.H. Sleight III. 1979. Considerations in assessing the potential for, and significance of, biomagnifications of chemical residues in aquatic food chains. In *Aquatic Toxicology*, *ASTM STP 667*. Marking, L.L., and R.A. Kimerle (eds.). American Society for Testing and Materials, Philedelphia. pp. 251-268.

Mackay, D., and S. Paterson. 1982. Fugacity revisited. *Environ. Sci. Technol.* 16:654A-660A and references therein.

Mahle, N.H., and L.F. Whiting. 1980. The formation of chlorodibenzo-p- dioxins by air oxidation and chlorination of bituminous coal. Dow Chemical Co., Midland, Michigan.

Manara, L., Coccia, P., and T. Croci. 1982. Persistent tissue levels of TCDD in the mouse and their reduction as related to prevention of toxicity. Drug Metab. Rev. 13(3):423-446.

Martinucci, G.B., Crespi, P., Omodeo, P., Osella, G., and G. Traldi. 1983. Earthworms and TCDD (2,3,7,8-tetrachlorodibenzo-p-dioxin) in Seveso. In Earthworm Ecology - From Darwin to Vermiculture. J.E. Satchell (ed.). Chapman and Hall, London. pp. 275-283.

Masuda, Y., and H. Yoshimura. 1984. Polychlorinated biphenyls and dibenzofurans in patients with Yusho and their toxicological significance: A review. Amer. J. Ind. Med. 5:31-44.

Matsumura, F., Tsushimota, G., and R. Sago. 1982. Fate of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in an outdoor pond and in model aquatic ecosystems. Environmental Toxicology and Chemistry 1:61-68.

Matsumura, F., and H.J. Benezet. 1973. Studies on the bioaccumulation and microbial degradation of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Environ. Health Perspect. 5:253-258.

Matsumura, F., Quensen, J., and G. Tsushimoto. 1983a. microbial degradation of TCDD in a model ecosystem. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L. and A.P. Gray (eds.). Plenum Press, New York. pp. 191-219.

Matsumura, F., Quensen, J., and G. Tsushimoto. 1983b. Microbial degradation of TCDD in a model ecosystem. In: Accidental Exposure to Dioxins: Human Health Aspects. F. Coulston, et al. (eds.). Academic Press, New York. pp. 105-135

Mayer, F.L., Mehrle, P.M., and H.O. Sanders. 1977. Residue dynamics and biological effects of polychlorinated biphenyls in aquatic organisms. Arch. Environ. Contam. Toxicol. 5:501-511.

May, G. 1982. Tetrachlorodibenzodioxin: A survey of subjects ten years after exposure. Br. J. Industr. Med. 39:128-135.

Mazer, T., Hileman, F.D., Noble, R.W., Hale, M.D., and J. Brooks. 1983. Characterization of tetrachlorodibenzofurans. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Boston. pp. 23-58.

McCague, G.R. 1978. No environmental health hazard from transformer fire. Ministry of Environment, Ontario, Canada. News Release, February 13, 1978.

McConnell, E.E., Lucier, G.W., Rumbaugh, R.C., Albro, P.W., Harvan, D.J., Hass, J.R., and M.W. Harris. 1984. Dioxin in soil: Bioavailability after ingestion by rats and guinea pigs. Science 223:1077-1079.

McConnell, E.E., Moore, J.A., Haseman, J.K., and M.W. Harris. 1978a. The comparative toxicity of chlorinated dibenzo-p-dioxins in mice and guinea pigs. Toxicol. Appl. Pharmacol. 44:335-356.

McConnell, E.E., Moore, J.A., and D.W. Dalgard. 1978b. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in Rhesus monkeys (Macaca mulatta) following a single oral dose. Toxicol. Appl. Pharmacol. 43:175-187.

McKinney, J.D., Chae, K., Gupta, B.N., Moore, J.A., and J.A. Goldstein. 1976. Toxicological assessment of hexachlorobiphenyl isomers and 2,3,7,8-tetrachlorodibenzofuran in chicks. 1. Relationship of chemical parameters. Toxicol. Appl. Pharmacol. 36:65-80.

McLaughlin, D.L., and R.G. Pearson. 1984. Concentrations of PCDD and PCDF in soil from the vicinity of the SWARU incinerator, Stoney Creek, July, 1983. Air Resources Branch, Ontario Ministry of the Environment. 22 pages.

McLeod, H.A. 1981. Personal communication to M.J. Boddington. Health Protection Branch, Health and Welfare Canada.

McMillin, C.R., Hileman, F.D., Kirk, D.E., Mazer, T., Warner, B.J., Wesselman, R.J., and J.E. Longbottom. 1983. Determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin in industrial and municipal wastewaters, method 613: Performance evaluation and preliminary method study results. In Chlorinated Dioxins and Dibenzofurans in the Total Environment, Choudhary, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Woburn, MA. pp. 181-192.

McNulty, W.P., Nielsen-Smith, K.A., and J.O. Lay. 1982. Persistence of TCDD in monkey adipose tissue. *Food Cosmet. Toxic.* 20:985-987.

McNulty, W.P. 1977. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin for Rhesus monkeys: Brief report. *Bull. Environ. Contam. Toxicol.* 18:108-109.

Miller, R.A., Norris, and C.L. Hawkes. 1973. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in aquatic organisms. *Environmental Health Perspectives* 5:177-186.

Ministry of Energy. 1983. *Ontario Energy Review*. Third edition, pp. 1-64.

Ministry of Natural Resources. 1980. Survey of Ontario's Resident and Non-resident Sport Fishermen.

Ministry of the Environment. 1983. Summary of dioxin analyses for the Elmira area - Presentation by west central region to Uniroyal Chemical, Elmira, Technical Steering Committee, March 22, 1983.

Ministry of the Environment. 1984. Dioxin soil analysis results - Presentation by west central region to Uniroyal Chemical, Elmira, Technical Steering Committee, April 3, 1984.

Ministry of the Environment and Ministry of Natural Resources. 1983. *Guide to eating Ontario sportfish*.

Monteriolo, S.C., S. DiDomenico, A., Silano, V., Viviano, G., and G. Zapponi. 1982. 2,3,7,8-TCDD levels and distribution in the environment at Seveso after the ICMESA accident on July 10, 1976. In *Chlorinated Dioxins and Related Compounds: Impact on the Environment*. Hutzinger, O., Fre , R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 127-136.

Moore, J.A. (ed.). 1973. Perspective on chlorinated dibenzodioxins and dibenzofurans. *Environmental Health Perspectives* 5:313 pp.

Mortelman, K., Haworth, S., Speck, W., and E. Zeiger. 1984. Mutagenicity testing of agent orange components and related chemicals. *Toxicol. Appl. Pharmacol.* 75:137-146.

Mottura, A., Tenchini, M.L., Crimaudo, C., DeCarli, L., Giorgi, R., Nuzzo, F., Zei, G., Fraccaro, M., Veneroni, P., Nicoletti, B., Paggini, L., Simoni, G., Dalpra, L., and P. Mocarelli. 1981. Evaluation of results of chromosome analysis on lymphocytes of TCDD exposed subjects after the Seveso accident. Mutat. Res. 85(4):238-239.

Mulcahy, M.T. 1980. Chromosome observation and "Agent Orange". Med. J. Aust. 2(10):573-574.

Murray, F.J., Smith, F.A., Nitschke, K.D., Humiston, C.G., Kociba, R.J., and B.A. Schwetz. 1979. Three generation reproduction study of rats given 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in the diet. Toxicol. Appl. Pharmacol. 50:241-252.

Nagayama, J., Kiyohara, D., Kuratsune, M., and Y. Masuda. 1983. Inductive effect on aryl hydrocarbon hydroxylase of chlorinated dibenzofurans and 2,3,7,8-T_4CDD in mice and human lymphoblastoid cells. American Chemical Society, Division of Environmental Chemistry. 186th National Meeting, Washington, D.C., Preprinted Extended Abstract 23:167-170.

Nagayama, J., Tokudome, S., Kuratsume, M., and Y. Masuda. 1980. Transfer of polychlorinated dibenzofurans to the fetuses and offspring of mice. Food Cosmet. Toxicol. 18:153-157.

Nash, R.G., and M.L. Beall, Jr. 1978. Environmental distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) applied with silvex to turf in microagrosystem. N.T.I.S. EPA-1AG-DG-0054, 27 pages.

National Academy of Sciences. 1977. Problems of risk estimation. In Drinking Water and Health. Washington, D.C., pp. 25-65.

National Research Council of Canada. 1983. 2,4-D: Some Current Issues. NRCC Publication No. 20647, 99 pages.

National Research Council of Canada. 1981a. Polychlorinated dibenzo-p-dioxins: Criteria for their effects on Man and his Environment. NRCC Publication No. 18574, 251 pages.

National Research Council of Canada. 1981b. Polychlorinated Dibenzo-p-dioxins: Limitations to the Current Analytical Techniques. NRCC Publication No. 18576, 172 pages.

National Research Council of Canada. 1978a. Phenoxy Herbicides - Their Effects on Environmental Quality. NRCC Publication No. 16075, 440 pages.

National Research Council of Canada. 1978b. Polchlorinated Biphenyls: Biological Criteria for an Assessment of Their Effects on Environmental Quality. NRCC Publication No. 16077, 172 pages.

National Research Council of Canada. 1982. Chlorinated Phenols: Criteria for Environmental Quality. NRCC Publication No. 18578, 191 pages.

National Research Council. 1983. Identifying and Estimating the Genetic Impact of Chemical Mutagens. National Academy Press, Washington, D.C., pp. 295.

National Toxicology Program. 1982a. Bioassay of 2,3,7,8-tetrachlorodibenzo-p-dioxin for possible carcinogenicity (gavage study). DHHS Publ. No. (NIH) 82-1765.

National Toxicology Program. 1982b. Bioassay of 2,3,7,8-tetrachlorodibenzo-p-dioxin for possible carcinogenicity (dermal study). DHHS Publ. No. (NIH) 80-1757.

Nebert, D.W., Jensen, N., Perry, J., and T. Oka. 1980. Associations between ornithine decarboxylase E.C.-4.1.1.17 induction and the A_h locus in mice treated with polycylic aromatic compounds. J. Biol. Chem. 255:6836-6842.

Nestrick, T.J., and L.L. Lamparski. 1982. Isomer-specific determination of chlorinated dioxins for assessment of formation and potential environmental emission from wood combustion. Anal. Chem. 54:2292-2299.

Nestrick, T.J., Lamparski, L.L., Crummett, W.B., and L.A. Shadoff. 1982. Comments on variations in concentrations of organic compounds including polychlorinated dibenzo-p-dioxins and polynuclear aromatic hydrocarbons in fly ash from a municipal incinerator. Anal. Chem. 54:824-825.

Neubert, D., Zens, P., Rothenwallner, A., and H.J. Merker. 1973. A survey of embryotoxic effects of TCDD in mammalian species. Environmental Health Perspectives 5:67-79.

New York State. 1979. Interagency Task Force on Hazardous Wastes. Draft Report on Hazardous Waste Disposal in Erie and Niagara Counties, New York.

Newton, M., and S.P. Synder. 1978. Exposure of forest herbivores to 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in areas sprayed with 2,4,5-T. Bull. Environ. Contam. Toxicol. 20:743-750.

Nisbet, I.C.T. 1981. The role of exposure assessment in risk estimation: Research needs. In *Health Risk Analysis*. C.R. Richmond *et al.* (eds.). The Franklin Institute Press, Philadelphia. pp. 419-429.

Nolan, R.J., Smith, F.A., and J.G. Hefner. 1979. Elimination and tissue distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in female guinea pigs following a single oral dose. *Toxicol. Appl. Pharmacol.* 48(1):A162.

Norris, L.A. 1981. The movement, persistence and fate of the phenoxy herbicides and TCDD in the forest. *Residue Reviews* 80:65-135.

Norris, L.A., Montgomery, M.L., and E.R. Johnson. 1977. The persistence of 2,4,5-T in a Pacific Northwest forest. *Weed Sci.* 25:417.

Nova Scotia Power Corporation. 1983. *Pentachlorophenol: Use in NSPC Pole Retreatment Program.* Report No. 83-EP-2. 47 pages and Appendices.

Olie, K., Berg, M.V.D., and O. Hutzinger. 1983. Formation and fate of PCDD and PCDF from combustion processes. *Chemosphere* 12:627-636.

Olie, K., Lustenhouwer, J.W.A., and O. Hutzinger. 1982. Polychlorinated dibenzo-p-dioxins and related compounds in incinerator effluents. In *Chlorinated Dioxins and Related Compounds: Impact on the Environment*. Hutzinger, O., Fre , R.W., Merian, E., and F. Pocchiari (eds.), Pergamon Press, Oxford, pp. 227-244.

Olie, K., Vermeulen, P.L., and O. Hutzinger. 1977. Chlorodibenzo-p-dioxins and chlorodibenzofurans are trace components of fly ash and flue gas of some municipal incinerators in The Netherlands. *Chemosphere* 6:455-459.

Olson, J.R., Gasiewicz, T.A., and R.A. Neal. 1980a. Tissue distribution, excretion, and metabolism of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in the Golden Syrian hamster. *Toxicol. Appl. Pharmacol.* 56:78-85.

Olson, J.R., Holscher, M.A., and R.A. Neal. 1980b. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the Golden hamster. *Toxicol. Appl. Pharmacol.* 55:67-78.

Olson, J.R., Gasiewicz, T.A., Geiger, L.E., and R.A. Neal. 1983. The metabolism of 2,3,7,8-tetrachlorodibenzo-p-dioxin in mammalian systems. In Accidental Exposure to Dioxins - Human Health Aspects. Coulston, F., and F. Pocchiari (eds.). Academic Press, New York. pp. 81-103.

Ontario Ministry of Agriculture and Food. Weed Control in Lawns and Gardens. Publication 529. 72 pages.

Ontario Pesticides Advisory Committee (Ministry of the Environment). 1979. A review of the use of 2,4,-D, other phenoxy herbicides, and picloram by Ontario government agencies. 13 pages.

Owens, I.S. 1977. Genetic regulation of UDP-glucuronosyltransferase induction by polycyclic aromatic hydrocarbons in mice. J. Biol. Chem. 252:2827-2833.

Ozvacic, V.M. 1983. Witness statements for the Ministry of the Environment before the Consolidated Hearings Board in the matter of an application for Victoria Hospital Energy from Waste Plant, Westminster Campus, London, Ontario, pp. 130-173.

Ozvacic, V.M. 1985. Personal Communication of unpublished data. Air Resources Branch, Ontario Ministry of the Environment.

Ozvacic, V., Wong, G., Tosine, H., Clement, R., Osborne, J., and S. Thorndyke. 1984a. Determination of chlorinated dibenzo-p-dioxins, chlorinated dibenzofurans, chlorinated biphenyls, chlorobenzenes and chlorophenols in air emissions and other process streams at SWARU in Hamilton. Joint Report by Air Resources and Laboratory Services and Applied Research Branches, M.O.E. and Ontario Research Foundation. Report No. ARB-02-84-ETRD.

Ozvacic, V., Wong, G., Tosine, H., Clement, R.E., and J. Osborne. 1984b. Emissions of chlorinated organics from two municipal incinerators in Ontario. Presented at 77th APCA Meeting. San Francisco. June.

O'Keefe, P., Meyer, C., Hilker, D., Aldous, K., Jelus-Tyror, B., Dillon, K., Donnelly, R., Horn, E., and R. Sloan. 1983. Analysis of 2,3,7,8-tetrachlorodibenzo-p-dioxin in Great Lakes fish. Chemosphere 12(3):325-332.

Parkinson, A., and S. Safe. 1981. Aryl hydrocarbon hydroxylase induction and its relationship to the toxicity of halogenated aryl hydrocarbons. Toxicol. Environ. Chem. Rev. 4:1.

Petty, J. D., Smith, L.M., Bergqvist, P.-A., Johnson, J.L., Stalling, D.L., and C. Rappe. 1983. Composition of polychlorinated dibenzofuran and dibenzo-p-dioxin residues in sediments of the Hudson and Housatonic Rivers. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhry, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Boston. pp. 203-208.

Piper, W.N., Rose, R.Q., and P.J. Gehring. 1973. Excretion and tissue distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the rat. Environ. Health Perspect. 5:241-244.

Pitot, H.C. 1983. Evaluation of the toxic and carcinogenic risks of environmental chemicals to human beings: Scientific, legal and risk-benefit considerations. In: Concepts in Cancer Medicine, S.B. Kahn (ed.), Grune and Stratton, Inc., pp. 101-118.

Pitot, H.C., Goldsworthy, H.A., Campbell, H.A., and A. Poland. 1980. Quantitative evaluation of the promotion by 2,3,7,8-tetrachlordibenzo-p-dioxin of hepatocarcinogenesis from diethylnitrosamine. Cancer Res. 40:3616-3620.

Plimmer, J.R., and U.I. Klingebiel. 1971. Riboflavin photosensitized oxidation of 2,4-dichlorophenol. Science 174:407-408.

Poiger, H., Buser, H.R., and C. Schlatter. 1984. The metabolism of 2,3,7,8-tetrachlorodibenzofuran in the rat. Chemosphere 13:351-357.

Poiger, H., and C. Schlatter. 1980. Influence of solvents and absorbants on dermal and intestinal absorption of TCDD. Food Cosmet. Toxicol. 18:477-482.

Poiger, H., and C. Schlatter. 1983. Animal toxicology of chlorinated dibenzo-p-dioxins. Chemosphere 12:453-462.

Poiger, H., Weber, H., and C. Schlatter. 1982. Special aspects of metabolism and kinetics of TCDD in dogs and rats. Assessment of toxicity and TCDD-metabolite(s) in guinea pigs. In Chlorinated Dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, N.Y. pp. 317-325.

Poland, A., Glover, E., and A.S. Kende. 1976. Stereospecific, high affinity binding of 2,3,7,8-tetrachlorodibenzo-p-dioxin by hepatic cytosol. *J. Biol. Chem.* 251:4936-4946.

Poland, A., and E. Glover. 1979. An estimate of the maximum *in vivo* covalent binding of 2,3,7,8-tetrachlorodibenzo- p-dioxin to rat liver protein ribosomal RNA and DNA. *Cancer Res.* 39(9):3341-3344.

Poland, A., and E. Glover. 1980. 2,3,7,8-tetrachlorodibenzo-p-dioxin: Segregation of toxicity with the A_h locus. *Molec. Pharmacol.* 17:86-94.

Poland, A., and J.C. Knutson. 1982. 2,3,7,8-tetrachlorodibenzo-p- dioxin and related halogenated aromatic hydrocarbons: Examination of the mechanisms of toxicity. *Ann. Rev. Pharmacol. Toxicol.* 22:517-524.

Poland, A., Palen, D., and E. Glover. 1982. Tumour promotion by TCDD in skin of HRS/J hairless mice. *Nature* 300:271-273.

Potter, C.L., Sipes, I.G., and D.H. Russell. 1984. Hypothyroxinemia and hypothermia in rats in response to 2,3,7,8-tetrachlorodibenzo-p-dioxin administration. *Toxicol. Appl. Pharmacol.* 69:89-95.

Puhvel, S.M., Ertl, D.C., and C.A. Lynberg. 1984. Increased epidermal transglutaminase activity following 2,3,7,8-tetrachlordibenzo-p-dioxin: *In vivo* and *in vitro* studies in mouse skin. *Toxicol. Appl. Pharmacol.* 73:42-27.

Quilliam, M. 1978. Ministry of the Environment, Internal Report.

R.L. Pyne Metallurgical Consultants Incorporated. 1984. Preliminary evaluation of precious metals extraction from incinerated sewage sludge. Prepared for Wastewater Technology Centre, Canada Centre for Inland Waters, Environment Canada. 70pp.

Rappe, C. 1984. Chemical analyses of adipose tissues. In *Public Health Risks of the Dioxins.* Lowrance, W.W. (ed.). The Rockefeller University, N.Y. pp. 57-61.

Rappe, C., Marklund, S., Buser, H.R., and H.P. Bosshardt. 1978. Formation of polychlorinated dibenzo-p-dioxins (PCDD's) and dibenzofurans (PCDF's) by burning or heating chlorophenates. *Chemosphere* 7:269-281.

Rappe, C., Buser, H.R., and H.P. Bosshardt. 1979a. Dioxins, dibenzofurans and other polyhalogenated aromatics: Production, use, formation and destruction. Ann. N.Y. Acad. Sci. 320:1-18.

Rappe, C., Buser, H.R., Kuroki, H., and Y. Masuda. 1979b. Identification of polychlorinated dibenzofurans (PCDFs) retained in patients with Yusho. Chemosphere 8:259-266.

Rappe, C., Buser, H.R., Stalling, D.L., Smith, L.M., and R.C. Dougherty. 1981. Identification of polychlorinated dibenzofurans in environmental samples. Nature 292:524-6.

Rappe, C., Marklund, S., Bergqvist, P.A., and M. Hansson. 1982. Polychlorinated dioxins (PCDD's), dibenzofurans (PCDF's) and other polynuclear aromatics (PCPNA's) formed during PCB fires. Chemica Scripta 20:56-61.

Rappe, C., Marklund, S., Bergqvist, P.A., and M. Hansson. 1983a. Polychlorinated dioxins, dibenzofurans and other polynuclear aromatics formed during incineration and PCB fires. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Boston. pp. 99-124.

Rappe, C., Markland, S., Nygren, M., and A. Gara. 1983b. Parameters for identification and confirmation in trace analyses of polychlorinated dibenzo-p-dioxins and dibenzofurans. In Chlorinated Dioxins and Dibenzofurans in the Total Environment, Choudhary, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Woburn, MA. pp. 259-272.

Rappe, C., Nygren, M., and G. Gustafsson. 1983c. Human exposure to polychlorinated dibenzo-p-dioxins and dibenzofurans. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, C., Keith, L.H. and C. Rappe (eds.). Butterworth Publishers, Toronto. pp. 355-365.

Redford, D., Haile, C.L., and R.M. Lucas. 1983. Emissions of PCDDs and PCDFs from combustion sources. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L. and A.P. Gray (eds.). Plenum Press, New York. pp. 143-152.

Reggiani, G. 1982. Toxicology of TCDD and related compounds: Observations in man. In Chlorinated Dioxins and related Compounds, Impact on the Environment. Hutzinger, O., Frei, R., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Toronto. pp. 463-493.

Reggiani, G. 1983. Toxicology of TCDD and related compounds: Observations in man. Chemosphere 12:463-475.

Riihimaki, V., Asp, S., Pukkala, E., and S. Hernberg. 1983. Mortality and cancer morbidity among chlorinated phenoxyacid applicators in Finland. Chemosphere 12:779-784.

Rizzardini, M., Romano, M., Tursi, F., Salmona, M., Vecchi, A., Sironi, M., Gizzi, F., Benfenati, E., Garattini, S., and R. Fanelli. 1983. Toxicological evaluation of urban waste incinerator emissions. Chemosphere 12:559-564.

Roberts, R.E., and B.J. Rogers. 1957. The effects of 2,4,5-T brush spray on turkeys. Poult. Sci. 36:703.

Robinson, D., and R.D. Smillie. 1977. Ontario Ministry of the Environment. OTC Report 7715.

Rogers, A.M., Anderson, M.E., and K.C. Back. 1982. Mutagenicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin and perfluorodecanoic acid in L-5178Y mouse lymphoma cells. Mutation Research 105:445-449.

Roller, N.F. 1979. Survey of pesticide use in Ontario, 1978. Ontario Ministry of Agriculture and Food, Economics Branch.

Rose, J.Q., Ramsey, J.C., Wentzler, T. H., Hummel, R.A., and P.J. Gehring. 1976. The fate of 2,3,7,8-tetrachlorodibenzo-p-dioxin following single and repeated oral doses to the rat. Toxicol. Appl. Pharmacol. 36(2):209-226.

Rozman K., Rozman, T., and H. Greim. 1984. Effect of thyroidectomy and thyroxine on 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) induced toxicity. Toxicol. Appl. Pharmacol. 72:372-376.

Rudling, L., Ahling, B., and G. Lofroth. 1980. Chemical and biological characterization of emissions from combustion of wood and wood-chips in a small central heating furnace and from combustion of wood in closed fireplace stoves. English summary of a report in Swedish, submitted to and published by the National Swedish Environment Protection Board; SNV PM 1331.

Ryan, J.J. 1983. Higher chlorinated dioxins implicated in the mortality of young pigs kept on a pentachlorophenol-treated wooden floor. Can. Vet. J. 24:72-75.

Ryan, J.J., and J.C. Pilon. 1982. Chlorinated dibenzodioxins and dibenzofurans in chicken litter and livers arising from penta chlorophenol contamination of wood shavings. In Chlorinated Dioxins and Related Compounds, Impact on the Environment. Hutzinger, O., Frei., R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 183-189.

Ryan, J.J., and D.T. Williams. 1983. Analysis of human fat tissue from the Great Lakes area for 2,3,7,8-tetrachlorodibenzo-p-dioxin and furan residues. American Chemical Society, Division of Environmental Chemistry, 186th National Meeting, Washington, D.C. Preprint of Extended Abstracts 23:157-158.

Ryan, J.J., Pilon, J.C., Conacher, H.B.S., and D. Firestone. 1983a. Interlaboratory study on determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin in fish. J. Assoc. Off. Anal. Chem. 66(3):700-707.

Ryan, J.J., Lau, P.-Y., Pilon, J.C., and D. Lewis. 1983b. 2,3,7,8-tetrachlorodibenzo-p-dioxin and 2,3,7,8-tetrachlorodibenzofuran residues in Great Lakes commercial and sport fish. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, G., Keith, L.H. and C. Rappe (eds.). Butterworth Publishers, Boston. pp. 87-98.

Ryan, J.J., Lizotte, R., and B.P.-Y. Lau. 1984a. Chlorinated dibenzo-p-dioxins and furans in Canadian human adipose tissue. 4th International Symposium on Chlorinated Dioxins and Related Compounds. Ottawa, October 16 - 18, 1984.

Ryan, J.J., Schecter, A., Lizotte, R., and W.-F. Sun. 1984b. Distribution of dioxins and furans in human autopsy tissues from the general population. 4th International Symposium on Chlorinated Dioxins and Related Compounds. Ottawa, Ontario, October 16 - 18, 1984. Poster 36.

Sawyer, T., Bandiera, S., Safe, S., Hutzinger, O., and K. Olie. 1983. Bioanalysis of polychlorinated dibenzofurans and dibenzo-p-dioxin mixtures in fly ash. Chemosphere 12:529-535.

Sawyer, T., and S. Safe. 1982. PCB isomers and congeners: induction of aryl hydrocarbon hydroxylase and ethoxyresorufin O-deethylase enzyme activities in Rat Hepatoma cells. Toxicol. Lett. 13:87-94.

Schantz, S.L., Barotti. D.A., and J.R. Allen. 1979. Toxicological effects produced in non-human primates chronically exposed to fifty parts per trillion 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD). Toxicol. Appl. Pharmacol. 48:A180.

Schecter, A., and J. Ryan. 1984. Chlorinated dibenzodioxins and dibenzofurans in human adipose tissue in exposed and control populations. 4th International Symposium on Chlorinated Dioxins and Related Compounds. Ottawa, October 16 - 18, 1984, Poster 37.

Schmidt-Bleek, F., Strachan, W.J.M., Klein, A.W., and J.M. Hushon. 1983. Use of OECD premarket data in environmental exposure analysis for new chemicals. Chemosphere 12:887.

Schoeny, R. 1982. Mutagenicity of chlorinated biphenyls and chlorinated dibenzofurans. Mutation Research 101:45-56.

Schwetz, B.A., Norris, J.M., Sparschu, G.L., Rowe, V.K., Gehring, P.J., Emerson, J.L., and C.G. Gerbig. 1973. Toxicology of chlorinated dibenzo-p-dioxins. Environmental Health Perspectives 5:87-99.

Seiler, J.P. 1973. A survey on the mutagenicity of various pesticides. Experientia 29:622.

Shadoff, L.A., et. al. 1977. The gas chromatographic-mass spectrometric determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin in fat from cattle fed Ronnel insecticide. Ann. Chem. 67:583-592.

Shaub, W.M., and W. Tsang. 1983. Dioxin formation in incinerators. Environ. Sci. Technol. 17:721-730.

Shields, J.K. 1976. Control of preservative wastes from wood treatment. Environment Canada. Report OPX 163E.

Silkworth, J., McMartin, D., DeCaprio, A., Rej, R., O'Keefe, P., and L. Kaminsky. 1982. Acute toxicity in guinea pigs and rabbits of soot from a polychlorinated biphenyl-containing transformer fire. Toxicol. Appl. Pharmacol. 65:425-439.

Smith, A.H., Fisher, D.O., Giles, H.J., and N. Pearce. 1983. The New Zealand soft tissue sarcoma case-control study: Interview findings concerning phenoxyacotic acid exposure. Chemosphere 12:565-571.

Smith, F.A., Schwetz, B.A., and K.D. Nitschke. 1976. Teratogenicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in CF-1 mice. Toxicol. Appl. Pharmacol. 38:517-523.

Smith, R. M., O'Keefe, P., Aldous, K.M., Hilker, D.R., and J.E. O'Brien. 1983. 2,3,7,8-tetrachlorodibenzo-p-dioxin in sediment samples from Love Canal storm sewers and creeks. Environ. Sci. Technol. 17(1):6-10.

Somers, E., and V.M. Douglas. 1983. Dioxins and related compounds as issues of international concern. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L., and A.P. Gray (eds.). Plenum Press, New York. pp. 33-41.

Sparschu, G.L., Dunn, F.L., and V.K. Rowe. 1971. Study of the teratogenicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin in the rat. Food Cosmet. Toxicol. 9:405-412.

Squire, R.A. 1981. Ranking animal carcinogens: a proposed regulatory approach. Science 214:877-880.

Stalling, D. L., Petty, J.D., Smith, L.M., Rappe, C., and H.R. Buser. 1982. Isolation and analysis of polychlorinated dibenzofurans in aquatic samples. In Chlorinated dioxins and Related Compounds: Impact on the Environment. Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 77-85.

Stalling, D.L., Smith, L.M., Petty, J.D., Hogan, J.W., Johnson, J.L., Rappe, C., and H.R. Buser. 1983a. Residues of polychlorinated dibenzo-p-dioxins and dibenzofurans in Laurentian Great Lakes fish. In Human and Environmental Risks of chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L., and A.P. Gray (eds.). Plenum Press, New York. pp. 221-240.

Stalling, D. L., Petty, J.D., Smith, L.M., and W.J. Dunn III. 1983b. Dioxins and furans in the environment - A problem for chemometrics. In Dioxins in the Environment. Michigan State University, East Lansing, Michigan. (In press).

Stanley, J.S., Going, J.E., Redford, D.P., Kutz, J.W., and A.L. Young. 1983. Analytical methods for measurement of polychlorinated dibenzo-p-dioxins (PCDDs) in human adipose tissues. American Chemical Society, Division of Environmental Chemistry, 186th National Meeting, Washington, D.C. Preprint of Abstracts 23:155-156.

Stevens, K.M. 1981. Agent orange toxicity: A quantitative perspective. Human Toxicology 1:31-39.

Steward, A.R., and Byard, J.L. 1981. Induction of benzo(a)pyrene metabolism by 2,3,7,8-tetrachlorodibenzo-p-dioxin in primary cultures of adult rat hepatocytes. Toxicol. Appl. Pharmacol. 59:603-616.

Sundstrom, G., Jensen, S., Jansson, B., and K. Erne. 1979. Chlorinated phenoxyacetic acid derivatives and tetrachlorodibenzo-p-dioxin in foliage after application of 2,4,5-trichlorophenoxyacetic acid esters. Arch. Environm. Contam. Toxicol. 8:441-448.

Swiss Federal Office for Environmental Protection. 1982. Environmental pollution caused by dioxins and furans from communal refuse incineration plants. Leaflet on Environmental Protection No. 5. Martin GmbH, Munchen, pp. 22.

Taylor, M.L., Tiernan, T.O., Garrett, J.H., VanNess, G.F., and J.G. Solch. 1983. Assessments of incineration processes as sources of supertoxic chlorinated hydrocarbons: Concentrations of polychlorinated dibenzo-p-dioxins/dibenzofurans and possible precursor compounds in incinerator effluents. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Boston. pp. 125-164.

Thalken, C.E., and A.L. Young. 1983. Long-term field studies of a rodent population continuously exposed to TCDD. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L., and A.P. Evoy (eds.). Plenum Press, New York. pp. 357-372.

Theiss, J.C. 1983. The ranking of chemicals for carcinogenic potency. Regulatory Toxicol. Pharmacol. 3:320-328.

Thibodeaux, L.J. 1983. Offsite transport of 2,3,7,8-tetrachlorodibenzo-p-dioxin from a production disposal facility. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Boston. pp. 75-86.

Tiernan, T.O. 1980. Direct Testimony of Dr. Thomas O. Tiernan. Before US EPA. In Re: The Dow Chemical Company, et al. Docket No. 415, et al. Exhibit 222.

Tiernan, T.O. 1983. Analytical chemistry of polychlorinated dibenzo-p-dioxins and dibenzofurans: A review of the current status. In Chlorinated Dioxins and Dibenzofurans in the Total Environment. Choudhary, G., Keith, L.H., and C. Rappe (eds.). Butterworth Publishers, Woburn, MA. pp. 211-237.

Tiernan, T.O., Solch, J.G., VanNess, G.F., Garrett, J.H., Wagel, D.J., Ferguson, J.L., and M.L. Taylor. 1983a. Analytical methodology for the determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin in soils. American Chemical Society, Division of Environmental Chemistry, 186th National Meeting, Washington, D.C., Preprint Extended Abstr. 23:215-217.

Tiernan, T.O., Taylor, M.L., Garrett, J.H., VanNess, G.F., Solch, J.G., Deis, D.A., and D.J. Wagel. 1983b. Chlorodibenzo-dioxins, chlorodibenzofurans and related compounds in the effluents from combustion processes. Chemosphere 12:595-606.

Tiernan, T.O., Taylor, M.L., Solch, J.G., VanNess, G.F., and J.H. Garrett. 1982a. Characterization of toxic components in the effluents from a refuse-fired incinerator. Resources and Conservation 9:343-354.

Tiernan, T.O., Taylor, M.L., Solch, J.G., VanNess, F., Garrett, J.H., and M.D. Porter. 1982b. Incineration of chemical wastes containing polychlorinated biphenyls: Assessment of tests conducted at Rollens Environmental Services, Deer Park, Texas, and Energy Systems Company, El Dorado, Arkansas. In Detoxication of Hazardous Waste. Exner, J.H. (ed.). Ann Arbor Science, Ann Arbor, Michigan. pp. 143-184.

Tosine, H., Clement, R.E., and R. Hunsinger. 1984. A Survey of Selected Drinking Water Supplies in Ontario for Chlorinated Dibenzo-p-dioxins and chlorinated Dibenzofurans. Laboratory Services and Applied Research Branch, Ontario Ministry of the Environment. 74 pages.

Tosine, H., Smillie, D., and G.A.V. Rees. 1983a. Comparative monitoring and analytical methodology for 2,3,7,8-TCDD in fish. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds, Tucker, R.E., Young, A.L., and A.P. Gray (eds.). Plenum Publishing Corp. pp. 127-139.

Tosine, H., Clement, R.E., Osborne, J., Ozvacic, V. and G. Wong. 1983b. Levels of chlorinated organics in a municipal incinerator. American Chemical Society, Division of Environmental Chemistry. 186th National Meeting, Washington, D.C. Preprinted Extended Abstract 23:260-262.

Townsend, D.I. 1980. The use of dioxin isomer group ratios to identify sources and define background levels of dioxins in the environment. J. Environ. Sci. Health B15:571-609.

Townsend, D.I. 1982. The use of dioxin isomer group ratios to identify sources and define background levels of dioxins in the environment. A Review, Update and Extension of the Present Theory. In Chlorinated Dioxins & Related Compounds Impact on the Environment, Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford and New York. pp. 265-274.

Townsend, D.I. 1983. Change of isomer ratio and fate of polychlorinated-p-dioxins in the environment. Chemosphere 12:637-643.

Tsang, W., and W. Shaub. 1982. Environmental consequences arising from the combustion of municipal solid waste. In Resource Recovery from Solid Wastes Proc. Conf. Sengupta, S., and K.F.V. Wong (eds.). Pergamon Press, New York. pp. 219-228.

Tsushimoto, G., Matsumura, F., and R. Sago. 1982, Fate of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in an outdoor pond and in model aquatic ecosystems. Environ. Toxicol. Chem. 1:61-68.

Ts'o, P.O.P. 1979. Current progress in the study of the basic mechanisms of neoplastic transformation. Banbury Rep. 2:385-406.

Tucker, R.K., and R.H. Hudson. 1970. 2,4,5-T acute and oral toxicity to mallard ducks. Private communication. November 15 to Denver Wildlife Research Centre, Colorado. U.S. Department of the Interior.

U.S. Fish & Wildlife Service, Columbia, Missouri, (1979).

U.S. Environmental Protection Agency. 1983. Dioxin Strategy. 43 pages.

U.S. Environmental Protection Agency. 1983. Assessment of emissions from a recent municipal waste combustor. Draft III.

U.S. Environmental Protection Agency. 1981. Interim evaluation of health risks associated with emissions of tetrachlorinated dioxins from municipal waste resource recovery facilities.

U.S.E.P.A. 1977. Dioxin Position Statement. Prepared by Dioxin Working Group. Washington, D.C. April 28, 1977.

Umbreit, T.H., Patel, D., and M.A. Gallo. 1984. Acute toxicity of TCDD contaminated soil from an industrial site. 4th International Symposium on Chlorinated Dioxins and Related Compounds. Ottawa, October 16 - 18, 1984. Poster 39.

van den Berg, M., Hutzinger, O., and K. Olie. 1983. Uptake and selective retention in rats of orally administered chlorinated dioxins and dibenzofurans from fly-ash and fly-ash extract. Chemosphere 12:537-544.

van den Berg, M., Olie, K., and O. Hutzinger. 1984. Bioavailability of fly ash-adsorbed PCDDs and PCDFs in the rat, guinea pig and Syrian golden hamster. 4th International Symposium on Chlorinated dioxins and Related Compounds. Ottawa, October 16 - 18, 1984. Poster 21.

Van Miller, J.P., Lalich, J.J., and J.R. Allen. 1977. Increased incidence of neoplasms in rats exposed to low levels of 2,3,7,8-tetrachlorodibenzo-p-dioxin. Chemosphere 6:625-632.

Van Miller, J.P., Marlor, R.J., and J.R. Allen. 1976. Tissue distribution and excretion of tritiated tetrachlorodibenzo-p-dioxin in non-human primates and rats. Food Cosmet. Toxicol. 14:31-34.

VanNess, G.F., Solch, J.G., Taylor, M.L., and T.O. Tiernan. 1980. Tetrachlorodibenzo-p-dioxins in chemical wastes, aqueous effluents and soils. Chemosphere 9:553-563.

Vaughan, B.E. 1983. Critical considerations in the assessment of health and environmental risks - what we have learned from the nuclear experience. The Science of the Total Environment 28:505-514.

Vecchi, A., Sironi, M., Canegratti, A., Recchia, M., and S. Garattini. 1983. Immunosuppressive effects of 2,3,7,8-tetrachloroibenzo-p-dioxin in strains of mice with different susceptibility to induction of aryl hydrocarbon hydroxylase. Toxic. Appl. Pharmacol. 68:34-441.

Verrett, M.J. 1970. Hearings before the Sub-Committee on Energy, Natural Resources, and the Environment of the Committee on Commerce, U.S. Senate, Serial 91-60, Government Printing Office, Washington, D.C.

Vos, J.G. 1978. 2,3,7,8-tetrachlorodibenzo-para-dioxin: Effects and mechanisms. In Chlorinated Phenoxy Acids and their Dioxins. C. Ramel (ed.). Ecological Bulletin No. 27. Swedish Natural Science Research Council. pp. 165-176.

Voyksner, R.D., Hass, J.R., Sovochool, G.W., and M.M. Bursey. 1983. Comparison of gas chromatography/high-resolution mass spectrometry and mass spectrometry/mass spectrometry for detection of polychlorinated biphenyls and tetrachlorodibenzofuran. Anal. Chem. 55:744-749.

Wang, D.K.W., Chiu, D.H., Leung, P.K., Thomas, R.S., and R.C. Lao. 1983. Sampling and analytical methodologies for PCDD's and PCDF's in incinerators and wood burning facilities. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L., and A.P. Gray (eds.). Plenum Press, New York. pp. 113-126.

Ward, C.T., and F. Matsumura. 1978. Fate of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in a model aquatic environment. Arch. Environm. Contam. Toxicol. 7:349-357.

Ward, D.E., McMahon, C.K., and R.W. Johansen. 1976. An update on particulate emissions from forest fires. Paper 67-2.2. Proc. Portland, Oreg. 15 pp.

Wassom, J.S., Huff, J.E., and N. Loprieno. 1978. A Review of the genetic toxicology of chlorinated dibenzo-p-dioxins. Mutat. Res. 47(3,4):141-160.

Weber, H., Poiger, H., and C. Schlatter. 1982. Fate of 2,3,7,8-tetrachlorodibenzo-o-dioxin metabolites from dogs and rats. Xenobiotica 12(6):353-357.

Weber, H., Lamb, J.C., Harris, M.W., and J.A. Moore. 1984. Teratogenicity of 2,3,7,8-tetrachlorodibenzofuran (TCDF) in mice. Toxicol. Lett. 20:183-188.

Weisburger, J.H., and G.M. Williams. 1980. Chemical carcinogens. In Toxicology: The Basic Science of Poisons. Doull, J.D., Klaassen, C.D. and M.O. Amdur (eds.). MacMillan and Co. pp. 84-138.

Whitehead, C.C., and R.J. Pettigrew. 1972. The subacute toxicity of 2,4-dichlorophenoxyacetic acid and 2,4,5-trichlorophenoxyacetic acid to chicks. Toxicol. Appli. Pharmacol. 21:348.

White, J., Pickett, E., and J. Robinson. 1981. Fuelwood consumption in Ontario. Ministry of Natural Resources, Ministry of Energy and Energy Ontario. pp. 1-63.

Williams, D.A. 1982. Identification of organic emissions resulting from the burning of ammunition boxes at CFB, Petawawa, Ontario. Pollution Measurement Division, APCD, EPS, Environment Canada. 118 pp.

Windsor Star. 27.1.1984. "Low Levels of Dioxin Found on Fighting Island".

Wipf, H.K., Homberger, E., Neuner, N., Ranalder, U.B., Vetter, W., and J.P. Vuilleumier. 1982. TCDD-levels in soil and plant samples from the Seveso area. In Chlorinated Dioxins and Related Compounds - Impact on the Environment. Hutzinger, O., Frei, R.W., Merian, E., and F. Pocchiari (eds.). Pergamon Press, Oxford. pp. 115-126.

Wipf, H.K., Homberger, E., Neuner, N., and F. Schenker. 1978. Field trials on photodegradation of TCDD on vegetation after spraying with vegetable oil. In Dioxin: Toxicological and Chemical Aspects. Cattabeni, F., Cavallaro, A., and G. Galli (eds.). Spectrum Publications Inc., New York. pp. 201-217.

Wong, A.S., Orbanosky, M.W., Taylor, P.A., McMillin, C.R., Noble, R.W., Wood, D., Longbottom, J.E., Forest, D.L., and R.J. Wesselman. 1983. Determination of 2,3,7,8-tetrachlorodibenzo-p-dioxin in industrial and municipal wastewaters, Method 613: Development and detection limits. In Chlorinated Dioxins and Dibenzofurans in the total Environment. Choudhary, G., Keith, L.H. and C. Rappe (eds.). Butterworth Publishers, Woburn, MA. pp. 165-180.

Yochim, R. S., Isensee, A.R., and G.E. Jone. 1978. Distribution and toxicity of TCDD and 2,4,5-T in an aquatic model ecosystem. Chemosphere 7(3):215-220.

Yoshihara, S., Nagata, K., Yoshimura, H., Kuroki, H., and Y. Masuda. 1981. Inductive effect on hepatic enzymes and acute toxicity of individual polychlorinated dibenzofuran congeners in rats. Toxicol. Appl. Pharmacol. 59:580-588.

Young, A.L. 1974. Ecological studies on a herbicide-equipment test area (TA C-52A), Eglin AFB Reservation, Florida. Air Force Technical Report AFATL-TR-74-12. N.T.I.S. AD-780-517, 146 p.

Young, A.L. 1983. Long-term studies on the persistence and movement of TCDD in a natural ecosystem. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R.E., Young, A.L., and A.P. Gray (eds.). Plenum Press, New York. pp. 173-190.

Young, A.L., and E.L. Arnold. 1983. Environmental fate of TCDD - Conclusion from three long-term field studies. American Chemical Society, Division of Environmental Chemisty, 186th National Meeting, Washington, D.C., Preprint Extended Abst. 23:87-90.

Young, A.L., Cairney, W.J., and C.E. Thalken. 1983. Persistence, movement and decontamination studies of TCDD in storage sites massively contaminated with phenoxy herbicides. Chemosphere 12:713-726.

Zack, J., and W. Gaffey. 1983. A mortality study of workers employed at the Monsanto Company plant in Nitro, West Virginia. In Human and Environmental Risks of Chlorinated Dioxins and Related Compounds. Tucker, R., Young, A., and A. Gray (eds.). Plenum Press, New York. pp. 575-591.

Zack, J.A., and R.R. Suskind. 1980. The mortality experience of workers exposed to tetrachlordibenzodioxin in a trichlorophenol accident. J. Occup. Med. 22:11-14.

Zitko, V., Wildish, D.J., Hutzinger, O., and P.M.K. Choi. 1973. Acute and chronic oral toxicity of chlorinated dibenzofurans to salmonid fishes. Environmental Health Perspectives 5: 187-189.

Zitko, V., and P.M.K. Choi. 1973. Oral toxicity of chlorinated dibenzofurans to juvenile Atlantic Salmon. Bull. Environ. Contam. Toxicol. 10:120-122.